THE FIRE ECOLOGY OF SAND SAGEBRUSH-MIXED PRAIRIE IN THE SOUTHERN GREAT PLAINS

by

LANCE THOMAS VERMEIRE, B.S., M.S.

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CHAPTER I

INTRODUCTION

The Great Plains evolved with the influences of fire, herbivory, and the interactions of these two factors. Following settlement, rangelands in the Great Plains typically continued to be grazed, but fire was generally suppressed. The ecological importance of fire in grassland ecosystems has since been recognized and reintroducing fire to these landscapes is gaining interest. Much is already known about the fire ecology of many habitats in the Great Plains, including mixed prairie. However, the ecological effects of fire and the interactions of fire and grazing are not well-documented for the sand sagebrush-mixed prairie habitat type.

In the Great Plains, sand sagebrush (*Artemisia filifolia* Torr.) is associated with mixed prairie, occasionally attains heights approaching 150 cm, and often accounts for 20 to 50% of the canopy cover. Therefore, understanding the fire ecology of the sand sagebrush-mixed prairie habitat type primarily demands knowledge about the ecology and physiology of sand sagebrush and its response to fire. Reports on sand sagebrush response to fire are limited in number, conflicting in their conclusions, and unsupported by data. Information on the sprouting ability, survival rates, and growth of sand sagebrush are necessary to assess the evolutionary and management implications of fire in sand sagebrush-mixed prairie. Knowledge of the total non-structural carbohydrate trends in sand sagebrush roots would also indicate likely periods of resistance and vulnerability to fire and other disturbances, including herbicides.

Fire effects may extend to the animals that utilize sand sagebrush-mixed prairie. The distribution of large herbivores is known to be affected by fire because of alterations in the structure, species composition, and forage quality of plant communities. Selective grazing of burned sites would be expected to alter vegetation beyond the effects of fire alone. Researchers have determined that grazers spend more time on burned sites, but herbage standing crop on and surrounding burned sites has not been measured to determine whether forage utilization changes in a predictable manner with distance from the burn. If the relationship between forage utilization and distance from burned sites is strong, prescribed fire may be effectively used as a grazing distribution tool and provide insight on landscape conditions prior to settlement and fire suppression.

Grasshoppers (Orthoptera: Acrididae) are among the most important seasonal food sources for wildlife in the Great Plains, particularly birds. However, some species of grasshoppers are considered very detrimental because of the damage they periodically inflict on rangelands and agricultural crops. Pesticides can effectively control grasshoppers, but are not species specific and broad-scale control may negatively affect non-target species. Fire has been shown to alter grasshopper communities, presumably through habitat alteration. However, it may be possible to prescribe fire for more specific control of grasshoppers based on the biology of individual species. If species-specific control can be achieved with prescribed fire, it may be possible to target pests while maintaining the food base for grasshopper predators.

Our general objectives were to examine the physiology of sand sagebrush and determine the effects of fall and spring prescribed fire on sand sagebrush, cattle grazing distribution and forage use, and grasshopper abundance and biomass within the sand sagebrush-mixed prairie habitat type of the southern Great Plains. The following chapter is a review of the literature addressing these objectives. The individual studies and specific objectives are presented in Chapters III through VI and are formatted for immediate submission to the *Journal of Range Management*.

CHAPTER II

LITERATURE REVIEW

Sand Sagebrush

Sand sagebrush (*Artemisia filifolia* Torr.) is an elliptical to globose-shaped shrub seldom exceeding 1m tall (Stubbendieck et al. 1997). The shrubs flower from July to October and are reported to reproduce primarily from seed. Each plant produces numerous achenes about 1 mm long that can remain on the shrubs through winter. Stems are brittle with exfoliating bark. Leaves are simple or 3-lobed, filiform, and occur in fascicles. The foliage has a bluish-green color when photosenthetically active and turns gray with senescence. The entire plant is aromatic with a strong sage smell. Forage quality is poor to fair for large herbivores. No clear trends have been shown between grazing intensity and sand sagebrush cover (Sims et al. 1976, Collins et al. 1987).

Sand sagebrush is distributed throughout the western Great Plains from South Dakota through Coahuila and from Oklahoma west to Nevada (Stubbendieck et al. 1997). Within this region, the late-seral shrub has a prominent presence on about 39-million ha of sandy rangelands (Klingman 1962). Sand sagebrush is adapted to well-drained soils of low fertility and gains dominance as soil texture becomes more sandy (Rasmussen and Brotherson 1984).

Sand sagebrush canopy cover can range from 20 to 50% in the southern Great Plains (Collins et al. 1987). At high densities, sand sagebrush reduces herbaceous plant production, foraging efficiency of large herbivores, and wildlife habitat suitability. Controlling sand sagebrush with 33% canopy cover has improved cattle performance and increased grazing capacity by 45% (McIlvain and Savage 1949). Much of the research on sand sagebrush has therefore focused on methods of control. The effects of chemical and mechanical treatments have been variable and generally short-lived.

Control of sand sagebrush with single applications of 2,4-D (2, 4-dichlorophenoxyacetic acid) has ranged from 30 to 90% (McIlvain and Savage 1949, Bovey 1964). Consistently achieving greater than 90% control has required applying 2,4-D at 1.1 to 2.2 kg ha^{-1} for 2 consecutive years (Bovey 1964, Wilson 1989). McIlvain and Savage (1949) and Bovey (1964) recommended applying herbicides when sand sagebrush was growing rapidly. Repeated mowing has reduced sand sagebrush as well, but McIlvain and Savage (1949) concluded mowing was not economically feasible and could not be applied over large tracts.

High levels of sand sagebrush control would generally be detrimental because the shrubs provide beneficial cover for the soil and some species of wildlife. Sandy soils are susceptible to severe wind erosion (Nance et al. 1960), but the canopy architecture of sand sagebrush has high potential for stabilizing soils exposed to strong winds (Hagen and Lyles 1988). Lesser prairie chicken (*Tympanuchus pallidicinctus*) density has been positively correlated with sand sagebrush when shrub cover was low to moderate (Cannon and Knopf 1981). Rodgers and Sexson (1990) also found avian diversity and abundance were reduced following chemical control of sand sagebrush in a 900-ha block. They recommended control efforts be limited to smaller patches within the landscape.

Management of species endemic to sand sagebrush habitat types may require maintenance of moderately dense sand sagebrush stands.

The fire ecology of many vegetation types in the Great Plains is well-documented (Wright and Bailey 1982). Yet, little is known about the ecological effects of fire on sand sagebrush, despite its ecological importance and the extent of contiguous stands. Sand sagebrush has been described as a sprouting species (Wright 1972, Wright and Bailey 1982) and a non-sprouting species that rapidly recolonizes burned sites with numerous seedlings (Wright and Bailey 1980). These conflicting accounts cite Jackson (1965) as the original source for both descriptions, but no data are presented by any of the sources.

Most *Artemisia* shrub species in the United States are intolerant of fire (Dix 1960, Rowe 1969, Wright and Bailey 1982). However, sprouting has been considered an evolutionary means of existence in ecosystems prone to fire and other frequent disturbances (James 1984, Kruger et al. 1997) and the southern Great Plains is believed to have had a 5 to 10-year natural fire frequency (Wright and Bailey 1982). Under such conditions, sand sagebrush would have to be a strong sprouter or produce numerous viable seeds and grow quickly. Otherwise plant density would have been relatively low.

Total Non-Structural Carbohydrates

Plant response to control measures or other disturbances depends strongly on timing relative to the plants' carbohydrate trends (Sosebee 1983). This is particularly true for plants capable of sprouting because roots must be killed to control the plant. Root-kill from foliar-applied systemic herbicides is greatest when roots are a strong sink for carbohydrates because translocation to the perennating organs is maximized (Fisher et al. 1956, Brady and Hall 1976, Fick and Sosebee 1981). Rapidly growing plant parts are typically a sink for carbohydrates, so control from herbicide applied during active stem growth is usually limited to top-kill (Sosebee 1983). Fire and mechanical treatments are most detrimental if damage to above-ground tissues is inflicted when root carbohydrates are lowest (Jones and Laude 1960, Willard and McKell 1978). This has been shown for other *Artemisia* species that had portions of the canopy clipped (Cook and Stoddart 1959, Wright 1970). Top removal deprives the plants of new carbohydrates and forces the depletion of those in below-ground tissues (Coyne et al. 1995). Plants with rapid reduction and replenishment of carbohydrates are considered difficult to control because the period of vulnerability is narrow (Menke and Trlica 1981).

Proper timing of treatments is therefore dependent on the ability to recognize carbohydrate trends. Plant phenology is strongly correlated with carbohydrate trends (Coyne and Cook 1970, Wilson et al. 1975, Menke and Trlica 1981). Sosebee (1983) indicated that plant phenology can be used more reliably than calendar dates for determining the timing of treatments because of this relationship. Carbohydrate concentration can be affected by environmental stress, but the absolute concentrations of carbohydrates are rarely as important as the relative concentration during the annual cycle and the direction of transport to the various tissues (Sosebee 1983).

Grazing Distribution

Animal selectivity and its effects on the distribution of their resource utilization is one of the central issues in rangeland management. Uniform animal distribution is sought to avoid having areas of over-utilized and under-utilized forage resources. However, forage utilization is infrequently uniform because combinations of biotic and abiotic characteristics are rarely homogeneous across the landscape and herbivores naturally have preferences for site conditions conducive to their needs (Bailey et al. 1996).

Understanding many of these preferences, herbivore distribution has been altered with strategic placement of attractants such as shade (McIlvain and Shoop 1971), nitrogen fertilizer (Hooper et al. 1969, Samuel et al. 1980), salt, and supplemental feeds (Martin and Ward 1973, Bailey and Welling 1999). Fencing and implementing specialized grazing systems have also made animal distribution more uniform by limiting choices available to the animals (Vallentine 1990). Bailey and Welling (1999) reported increased forage use around dehydrated molasses was focused within 200 m of the supplement, but utilization was relatively uniform in the area affected.

Slope and distance to water are the overriding factors controlling distribution of forage use (Bailey et al. 1996). Because of the energy required to ascend steep slopes, cattle generally avoid them (Mueggler 1965, Cook 1966). Forage use around water has been shown in two patterns. Utilization is either greatest within 200 to 800 m of water then stabilizes (Gillen et al. 1984, Fusco et al. 1994), or decreases gradually with distance from water until sites become avoided at about 1600 m from water (Valentine 1947, Martin and Ward 1970).

Fire is a powerful tool that can affect animal distribution at various scales. Grazing distribution is often more uniform on burned pastures because differences in forage nutritive value, palatability, and accessibility among patches are reduced (Wright 1974) making selective patch grazing less likely. Given the choice of burned or non-burned sites, large herbivores strongly select burned sites as long as forage quantity is adequate (Coppock and Detling 1986, Mitchell and Villalobos 1999). Recommendations have therefore been made to use prescribed burning only on a management-unit basis. Wright (1974) suggested burned patches should be protected by fencing, or the remainder of the unit should be burned to prevent heavy localized grazing and overuse.

The Great Plains evolved with the interacting forces of fire and herbivory by large grazers (Axelrod 1985). Fuhlendorf and Engle (2001) proposed that patch burning could be used to increase heterogeneity across the landscape and would more closely approximate pre-settlement interactions of fire and grazing. Bison (*Bison bison* L.) have been shown to selectively graze burned patches (Coppock and Detling 1986) and preference among patches burned in spring, summer, and fall (Coppedge and Shaw 1998). Selective use of burned patches altered plant community composition (Coppedge and Shaw 1998) as Wright (1974) warned. However, grasses regained dominance within 2 or 3 years after patches were burned and grazed by bison in tallgrass prairie (Coppedge et al. 1998).

The fear of localized overuse of forages has prevented fire from being used to its potential as a distribution tool. If fire effects on distribution of forage use are strong and predictable, patch burning could be used to increase the uniformity of forage use, draw animals away from sensitive areas, or create greater landscape heterogeneity by encouraging concentrated forage use.

Grasshoppers

Grasshoppers (Orthoptera: Acrididae) are considered among the most detrimental invertebrate pests throughout the world because of the damage they inflict on agricultural crops and forage resources (Watts et al. 1989). Forage consumption rates of individual grasshoppers are relatively small, but they waste nearly as much as they consume and at high densities, can substantially reduce forage availability for livestock and herbivorous wildlife. Hewitt and Onsager (1983) estimated that more than 20% of the forage in the western United States is lost to grasshoppers annually.

Control efforts with insecticides have been controversial because of the costs, short treatment life, and potential effects on non-target species (Blickenstaff et al. 1974, Watts et al. 1989). Although grasshoppers compete with other herbivores and some species are agricultural pests, they also provide an important seasonal food source for many species of wildlife, particularly birds. Insecticides used for grasshopper control have had few direct effects on birds (McEwen et al. 1972, Stromborg et al. 1984, George et al. 1995). However, with grasshopper crude protein concentrations of 50 to 70% (Ueckert et al. 1972, DeFoliart 1975), predators could have difficulty overcoming their absence following large-scale control. Grasshoppers are a large component of summer and fall diets for gallinaceous birds (Davis et al. 1975, Doerr and Guthery 1983) and along with other insects, have been considered necessary for chick survival and growth (Johnson and Boyce 1990).

Of the 600 grasshopper species in the United States, only about a dozen frequently occur at high densities. Grasshopper abundance in the southern Great Plains has been reported to be strongly favored by dry years and heavy grazing (Smith 1940, Campbell et al. 1974). Selective control of these destructive species would maintain the food base for insectivores while protecting the forage base for herbivores. Fire is a natural phenomenon that has been shown to affect grasshopper assemblages, presumably through alterations in vegetative structure and species composition (Evans 1984, 1988). Prescribed fire is therefore a potential alternative to chemical control that may allow more specific targeting of pest species.

Direct mortality is likely if burns are conducted when grasshoppers are in a wingless nymph stage, or if eggs are exposed to lethal temperatures. Most grasshoppers lay eggs deeply in the soil and nymphs do not hatch until late April or early May (Pfadt 1988). However, variations in grasshopper hatching dates and methods of egg deposition may allow fire prescriptions to be directed at species-specific control. Species that overwinter as nymphs, such as *Eritettix simplex* (Scudder) and *Xanthippus corallipes* (Haldeman), could be specifically targeted by winter burns because the nymphs are relatively immobile and other species are protected in the soil as eggs. *Ageneotettix deorum* (Scudder) and *Aulocara elliotti* (Thomas) lay their eggs horizontally near the soil surface (Pfadt 1988) and may be vulnerable to the heat of passing fire. Dysart (1996) ranked *Aulocara elliotti* and *Ageneotettix deorum* the second and fifth most detrimental grasshoppers on Western rangelands, respectively. Both of these species are found in sand sagebrush-mixed prairie.

Conclusions

Sand sagebrush-mixed prairie has evolved with the influences of fire and grazing, yet little is known about the fire ecology of this vegetation type or the effects fire has on the dominant grazers occurring there. Our objectives were to: determine annual total nonstructural carbohydrate (TNC) trends for sand sagebrush and develop management recommendations based on the shrubs' developmental and physiological status; examine the ecological effects of fire on sand sagebrush by determining survival and sprouting ability following fall and spring prescribed fire, determine plant size relationships with survival, and describe the recovery and growth patterns of sand sagebrush canopies; determine cattle grazing preference for burned sites relative to non-burned sites, examine whether forage utilization was affected by season of burn, determine forb response to patch burning, and describe the relationship between forage utilization and distance from burned sites; and evaluate the effects of fall and spring prescribed burning on the abundance and biomass of grasshoppers and determine if species could be selectively controlled with prescribed fire.

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CHAPTER III

TOTAL NONSTRUCTURAL CARBOHYDRATE TRENDS OF SAND SAGEBRUSH AND MANAGEMENT IMPLICATIONS

Abstract

Sand sagebrush (*Artemisia filifolia* Torr.) is a dominant shrub on about 39 million ha of sandy rangelands in the Great Plains and greatly reduces grazing capacity. Results of chemical control have been variable, mowing has not been cost-effective, and little is known about the fire ecology of sand sagebrush. Plant response to chemical, mechanical, and fire treatment is typically related to plant development and physiological status at the time of treatment. The objectives were to determine the annual total non-structural carbohydrate (TNC) trends of sand sagebrush and indicate likely periods of susceptibility and resistance to control and disturbance. Roots were collected from 10 shrubs each month and shrub phenology was recorded. Percent TNC was measured spectrophotometrically with an anthrone reagent and glucose standard. Peak root TNC concentration occurred from late flower development in September through senescence in November. Concentrations declined after leaf fall and stabilized until stem elongation. Root TNC was lowest during May and June, when stem elongation was greatest. Fire and mechanical treatments should be most detrimental when applied during this period. Root TNC accumulation rates were greatest during early flowering in July. The early

flowering stage following stem elongation would be the best time for active transport of herbicides to the root system, likely maximizing sand sagebrush mortality.

Introduction

Sand sagebrush (*Artemisia filifolia* Torr.) is a late-successional shrub on sandy soils from South Dakota through northern Mexico and from eastern Nevada through western Oklahoma. Sand sagebrush helps stabilize soils that are susceptible to wind erosion (Hagen and Lyles 1988) and provides important cover for some wildlife (Rodgers and Sexson 1990). Lesser prairie chicken (*Tympanuchus pallidicinctus*) density, for example, has been positively correlated with sand sagebrush when shrub cover was low to moderate (Cannon and Knopf 1981). Management of species such as the lesser prairie chicken may require maintenance of moderately dense sand sagebrush stands. However, sand sagebrush often dominates the landscape, reducing herbaceous forage yields, foraging efficiency of large herbivores, and wildlife habitat suitability. Controlling sand sagebrush has improved cattle performance and increased grazing capacity by 45% (McIlvain and Savage 1949). Whether management goals call for stand maintenance or reduction, the ability to predict sand sagebrush resistance and susceptibility to management practices is imperative.

The effects of chemical and mechanical treatments have been variable and generally short-lived. Control of sand sagebrush with single applications of 2,4-D (2, 4-dichlorophenoxyacetic acid) has ranged from 30 to 90% (McIlvain and Savage 1949, Bovey 1964). Consistently achieving greater than 90% control has required applying 2,4-

D at 1.1 to 2.2 kg ha^{-1} for 2 consecutive years (Bovey 1964, Wilson 1989). Repeated mowing has reduced sand sagebrush as well, but McIlvain and Savage (1949) concluded mowing was not economically feasible and could not be applied over large tracts. Although the fire ecology of sand sagebrush is not well-understood, the shrub appears to be a strong sprouter following fire in the Southern Great Plains (Vermeire et al. 2001).

The response of sprouting plants to control measures or other disturbances depends strongly on the timing of the disturbance relative to the plants' carbohydrate trends. Root-kill from foliar-applied systemic herbicides is greatest when roots are a strong sink for carbohydrates because translocation to the perennating organs is maximized (Fisher et al. 1956, Brady and Hall 1976, Fick and Sosebee 1981). Fire and mechanical treatments are most detrimental if damage to above-ground tissues is inflicted when root carbohydrates are lowest, depriving the plants of new carbohydrates and forcing the depletion of those in below-ground tissues (Coyne et al. 1995). Proper timing of treatments is therefore dependent on the ability to recognize carbohydrate trends.

Plant phenology is often correlated with carbohydrate trends and is more reliably used to recommend timing of treatments than calendar dates (Sosebee 1983). Our objectives were to determine annual total nonstructural carbohydrate (TNC) trends for sand sagebrush and develop management recommendations based on the shrubs' developmental and physiological status. We tested the hypotheses that root TNC concentration and trends vary among annual stages of shrub development.

Methods and Materials

Study Area

The study was conducted in northwestern Oklahoma on the Hal and Fern Cooper Wildlife Management Area, about 15 km northwest of Woodward (36° 34' N, 99° 34' W, elev. 625 m). The area consists of high-seral sandhills vegetation of the sagebrush-bluestem vegetation type (Küchler 1964). The climate is continental with a mean annual precipitation of 572 mm, 70% of which occurs as rain during the growing season (Apr-Sep). Mean monthly temperatures range from 1° C in January to 29° C in July (Unpublished data, USDA-ARS).

Data were collected on Deep Sand ecological sites with slopes of 1 to 12%. Pratt loamy fine sands (sandy, mixed, mesic Lamellic Haplustalfs) dominate and are interspersed with Tivoli fine sands (mixed, thermic Typic Ustipsamments) on the tops of dunes (Nance et al. 1960). Sand sagebrush was the dominant woody plant, with scattered eastern redcedar (*Juniperus virginiana* L.) trees and isolated sand plum (*Prunus angustifolia* Marsh.) thickets. The herbaceous component was dominated by little bluestem [*Schizachyrium scoparium* (Michx.) Nash], gramas (*Bouteloua spp.* Lag.), western ragweed (*Ambrosia psilostachya* D.C.), sand bluestem (*Andropogon hallii* Hack.), and sand lovegrass [*Eragrostis trichodes* (Nutt.) Wood].

Methods

Ten mature sand sagebrush plants (80-100 cm tall) were selected each month from November 1999 through November 2000 and during the 2001 growing season.

Phenological stage of development was recorded for each shrub. The shrubs were excavated and a section of live root exceeding 5 g was cut and collected from below the root crown to determine TNC concentration. Roots were cleaned of soil and bark with a knife and stored on ice before being air-dried to a constant weight at 53° C. Dry samples were ground in a Wiley mill to pass a 40-mesh (0.5 mm) screen and were stored in air-tight bottles prior to chemical analysis. Two 500.0 mg subsamples were collected from each sample and boiled in hydrochloric acid before the filtrate was mixed and heated with an anthrone reagent as described by Conway et al. (1999). Root TNC concentrations were determined spectrophotometrically with a Cary 50 spectrophotometer (at 612 nm) by regressing sample absorbance on that of a glucose standard.

Monthly root TNC data were grouped into phenological stages and analyzed as a completely randomized design using analysis-of-variance (SAS Institute 1985). When differences occurred, developmental stage TNC means were separated using Fisher's Protected Least Significant Difference (Steele and Torrie 1980). An alpha level of 0.05 was used for all tests. Precipitation and mean daily temperature for sampling periods were calculated from local weather station data (Unpublished data, USDA-ARS) and plotted with TNC against time.

Results and Discussion

Annual precipitation was similar to the 62-year mean during both years of the study, but growing-season precipitation was 9 and 26% below average for 2000 and 2001, respectively (Fig. 3.1). Root TNC concentrations were lower in 2001 than 2000

($P<0.05$) and remained depressed for a longer period (Fig. 3.2). These differences were probably weather induced. The number and magnitude of root TNC recharging events as well as TNC concentration have been shown to be affected by environmental stress, such as drought, in other plants (Sosebee 1983). The sharp decline and recovery of root TNC in 2000 coincided with favorable precipitation and rapid canopy growth.

Although TNC concentration differed between years, the distinction was somewhat trivial. Management decisions should be based on root TNC concentrations relative to other periods within a year and the direction of TNC translocation. Root TNC trends were consistent between years when based on phenological stages of shrub development. Root TNC concentrations were greatest at flower maturity and early senescence (Table 3.1). Concentrations decreased during winter dormancy and remained stable through bud break and early leaf development. Root TNC declined most sharply and reached its lowest concentration when the rate of stem elongation was greatest, in May and June. The most rapid accumulation of root TNC occurred when leaves were well-developed, stem elongation had nearly ceased, and the shrubs were flowering. Roots continued accumulating TNC until reaching a plateau in the autumn.

Sand sagebrush displayed a narrow V-shaped TNC cycle when precipitation was near-average (Fig. 3.2). Menke and Trlica (1981) considered such patterns indicative of plants resistant to top removal because root reserves were only reduced for a short time. Similarly, the rapid replenishment of root TNC in narrow V-shaped cycles limits the window of opportunity for concentrated herbicide translocation to the roots. Inconsistent results of previous control efforts could be explained in large part by timing of treatment applications relative to TNC trends.

Our trend data indicate very rapid changes in root TNC concentrations during late spring and early summer if growing conditions are favorable. Sand sagebrush will be most susceptible to control during these periods. McIlvain and Savage (1949) and Bovey (1964) recommended applying herbicides when sand sagebrush was growing rapidly. However, our data indicate declining root TNC concentrations at this time (Table 3.1). Herbicide would be translocated along with carbohydrates to actively growing above-ground tissues, limiting control to top-kill (Sosebee 1983). Flowers can also be a strong carbohydrate sink and post-flower or post-fruit herbicide application is recommended for some species (Sosebee 1983). Sand sagebrush flowers from July through the first frost in the Southern Great Plains. However, root TNC reserves of sand sagebrush do not appear to be utilized for flowering or increased after flowers have matured. Foliar-applied herbicides most effectively control sprouting shrubs when nutrients, such as carbohydrates, are actively being transported to the root system (Fisher et al. 1956, Brady and Hall 1976, Fick and Sosebee 1981, Sosebee and Dahl 1991). Root TNC concentrations increased rapidly in July, indicating a downward transport of nutrients following the cessation of stem elongation and the initiation of flowering. This should be optimal timing for maximizing herbicide translocation to roots and may allow effective control with reduced herbicide rates. Wilson (1989) obtained 94% control of sand sagebrush with a single application of 2,4-D during the early flowering stage.

Mechanical treatments and prescribed burning should be most damaging to sand sagebrush if applied when stems are rapidly elongating because root TNC concentrations are lowest. Nearly all stem elongation occurred from 15 May through 29 June (Vermeire 2002), but could vary among years. Prescribed fire may not be a viable treatment option during this period in the Southern Great Plains. Most of the vegetation is green and actively growing during May and June, so fire may not carry well unless a sufficient amount of dead fuels have been accumulated. Additionally, recovery of warm-season herbaceous plants may be delayed because growth rates are greatly reduced by July (Gillen and McNew 1987). Burning conditions and herbaceous plant response will generally improve earlier in the spring. The second most vulnerable period for sand sagebrush is shortly after buds break and leaves begin to develop. If fire is an integral part of the management system and the maintenance of sand sagebrush stands is desired, resistance to fire damage should be greatest from maturity through dormancy. However, prescribed fire could be applied any time fuel conditions were favorable and stems were not elongating. Burns should be timed to allow rapid recovery of vegetation to avoid leaving highly erodible soils exposed for long periods.

Management Implications

Whether management goals are for the maintenance or control of sand sagebrush, the shrub's response to management is important because of its prominence and influence on sandy soils throughout the Great Plains. Wildlife managers have shown some concern about indirect effects on sand sagebrush from practices such as prescribed burning.

However, the TNC trends and sprouting ability of sand sagebrush offer little opportunity for control without multiple treatments being applied during stem elongation. Prescribed fire or mechanical treatments applied to the canopy from maturity through dormancy should allow removal of decadent shrub material with little or no damage to sand sagebrush stands. I believe the variable results of control efforts have been caused by the timing of treatments relative to TNC trends. Although periods of rapid canopy growth have been recommended for treating sand sagebrush with foliar-applied herbicides (McIlvain and Savage 1949, Bovey 1964), our results indicate herbicide translocation to the roots should be greatest when stem growth has stalled and flowering has been initiated. Coordinating herbicide treatment with downward TNC trends should produce better root-kill and allow lower application rates.

Calendar dates have not been reliable predictors of shrub development or TNC trends because both are affected by growing conditions and will vary among years. Relationships between plant development and root TNC trends were consistent for sand sagebrush and have been with other species as well (Coyne and Cook 1970, Wilson et al. 1975, Menke and Trlica 1981). I recommend monitoring shrub development to recognize TNC trends and properly time management actions for optimal results.

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Table 3.1. Total non-structural carbohydrates (TNC) in sand sagebrush roots by phenological stage of development and calendar date near Woodward, Okla. from Nov. 1999 through Sep. 2001.

Phenological Stage	Sample Dates	TNC	
		Mean	Standard error
		(%)	
Early leaf senescence	16 Nov.	37.96 b [1]	1.75
Leaf fall, dormancy	23 Dec., 20 Jan.	32.73 cd	1.75
Buds swollen	23 Feb.	34.90 bcd	2.48
Early leaf development	30 Mar., 26 Apr.	30.22 d	1.24
Stem elongation, full leaf	15 May, 29 Jun.	22.37 e	1.24
Flowering	24 Jul., 9 Sep.	36.95 bc	1.43
Maturity, early leaf senescence	5 Oct., 14 Nov.	48.39 a	1.75

[1] Means followed by different letters are significantly different (P<0.05).

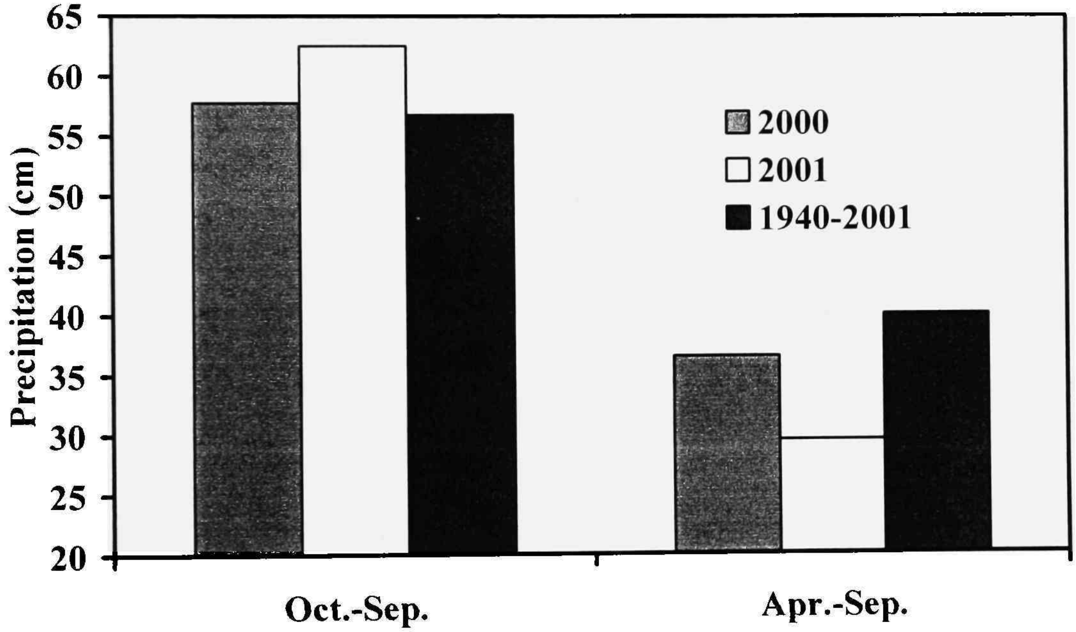

Figure 3.1. Annual (Oct.-Sep.) and growing-season (Apr.-Sep.) precipitation for Woodward, Okla. in 2000, 2001, and the 62-year mean from 1940 through 2001.

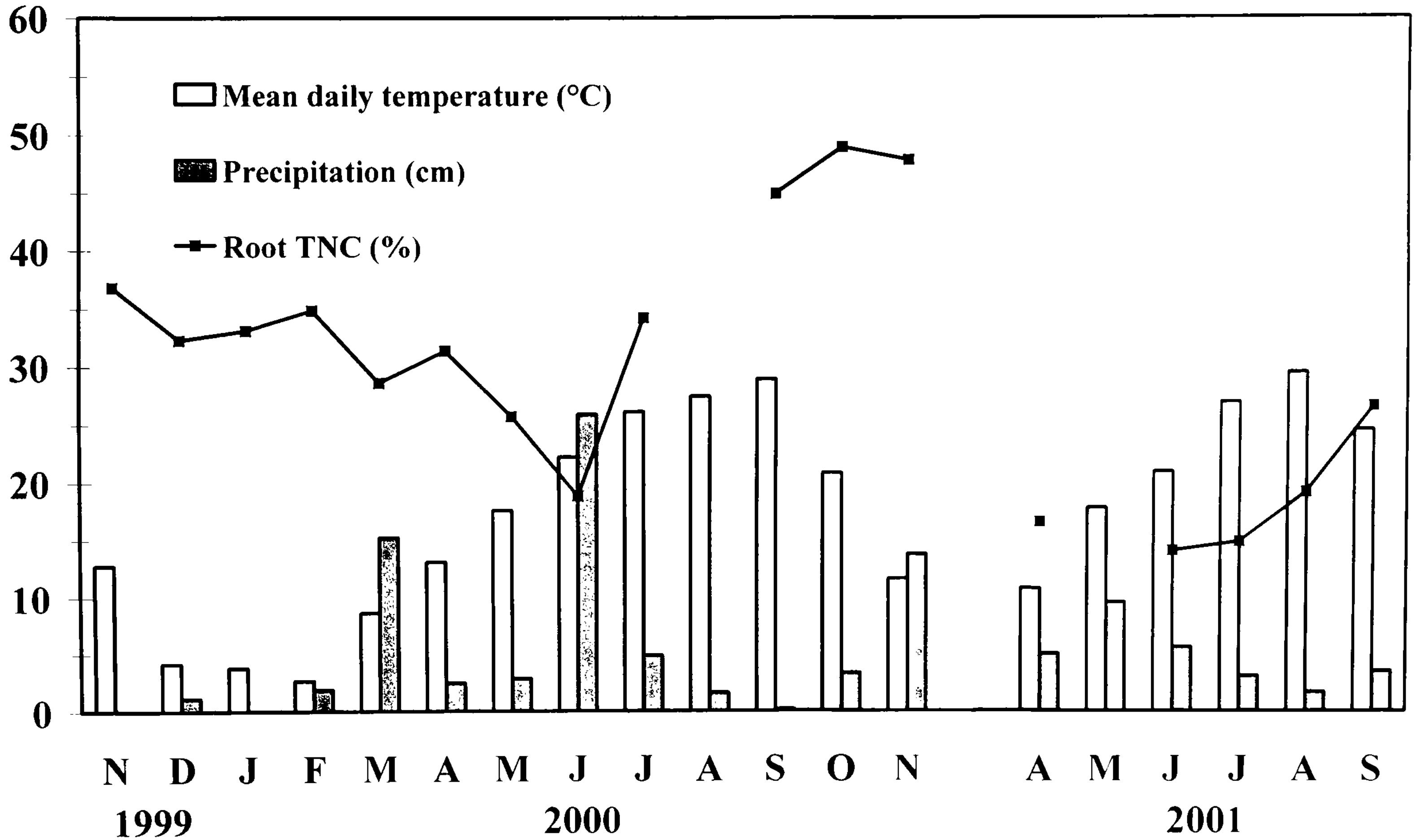

Figure 3.2. Total non-structural carbohydrate trends for sand sagebrush near Woodward, Okla. based on mean monthly root samples with mean daily temperature and cumulative precipitation for sampling periods (Nov. 1999-Sep. 2001).

CHAPTER IV

SEASONAL PRESCRIBED FIRE EFFECTS ON SAND SAGEBRUSH SURVIVAL AND GROWTH

Abstract

Sand sagebrush (*Artemisia filifolia* Torr.) is a dominant plant on sandy soils throughout the Great Plains and Southwest. Despite the ecological importance of sand sagebrush and the estimated 5- to 10-year natural fire frequency in the Great Plains, little is known about the ecological effects of fire on this species. Our objectives were to determine survival and sprouting ability of sand sagebrush following fall and spring prescribed fire, examine plant size relationships with survival, and describe the recovery and growth patterns of sand sagebrush canopies. I selected 24, 4-ha sites and assigned each site a fall, spring, or non-burned fire treatment with 4 replicates for each of 2 years. Twenty live shrubs were randomly selected in each plot, permanently marked with a rebar stake, and canopy measurements were made prior to treatment. Shrubs were monitored monthly from May through September to assess survival and growth in canopy height, area, and volume. Plots burned in the first year were also examined in September of the second year to observe additional growth and mortality after a second growing season. All non-burned shrubs survived during the study period and 96% of the burned shrubs survived. Delayed mortality was not a significant factor. Sprouting was positively correlated with shrub size. Burning in either season reduced canopy volume

64 and 38% below pre-treatment measurements in the first and second growing seasons after treatment. Sand sagebrush is a strong sprouter, but recovery rates indicate short fire return intervals would maintain the species in a less dominant phase than currently exists in areas deprived of fire.

Introduction

Sand sagebrush (*Artemisia filifolia* Torr.) is a dominant shrub on sandy soils from South Dakota through northern Mexico and from eastern Nevada through western Oklahoma. Klingman (1962) estimated sand sagebrush had a prominent presence on about 39-million ha of rangelands in the United States. At high densities, sand sagebrush can reduce herbaceous plant production, foraging efficiency of large herbivores, and wildlife habitat suitability. Much of the research on sand sagebrush has therefore focused on methods of controlling it (McIlvain and Savage 1949, Bovey 1964, Wilson 1989). However, high levels of sand sagebrush control could be considered detrimental because of its potential for reducing wind erosion (Hagen and Lyles 1988) and providing cover for some wildlife (Cannon and Knopf 1981, Rodgers and Sexson 1990).

Despite the ecological importance of sand sagebrush and the estimated 5 to 10-year natural fire frequency in the Great Plains (Wright and Bailey 1982), little is known about the ecological effects of fire on this species. Sand sagebrush has been described as a sprouting species (Wright 1972, Wright and Bailey 1982) and a non-sprouting species that recolonizes burned sites with seedlings (Wright and Bailey 1980). These conflicting accounts cite Jackson (1965) as the original source for both descriptions, but no data are presented by any of the sources. Our objective was to examine the ecological effects of fire on sand sagebrush by determining survival and sprouting ability following fall and spring prescribed fire, examining plant size relationships with survival, and describing the recovery and growth patterns of sand sagebrush canopies.

Methods and Materials

Study Area

The study was conducted in high-seral mixed prairie on the Hal and Fern Cooper Wildlife Management Area, about 15 km northwest of Woodward, Okla. (36° 34' N, 99° 34' W, elev. 625 m). The area consists of sandhills with slopes of 1 to 12%. The mean annual precipitation is 572 mm, with about 70% occurring as rain during the April through September growing season. Mean monthly temperatures range from 1° C in January to 29° C in July (Unpublished data, USDA-ARS). The area is lightly stocked with cow-calf and stocker cattle herds at 22 AUD ha^{-1} from early April to September.

Data were collected on Deep Sand ecological sites. The dominant soils were Pratt loamy fine sands (sandy, mixed, mesic Lamellic Haplustalfs) on interdunal sites and Tivoli fine sands (mixed, thermic Typic Ustipsamments) on the tops of dunes (Nance et al. 1960). Sand sagebrush (*Artemisia filifolia* Torr.) was the dominant woody plant, providing 20 to 50% canopy cover over most of the area. Sand plum (*Prunus angustifolia* Marsh.) and eastern redcedar (*Juniperus virginiana* L.) were sparsely distributed throughout the pastures. The herbaceous component was dominated by little bluestem [*Schizachyrium scoparium* (Michx.) Nash], gramas (*Bouteloua spp.* Lag.), western ragweed (*Ambrosia psilostachya* D.C.), sand bluestem (*Andropogon hallii* Hack.), sand lovegrass [*Eragrostis trichodes* (Nutt.) Wood], and Texas croton [*Croton texensis* (Kl.) Muell. Arg.].

Methods

I selected 24, 4-ha sites that were similar in vegetative composition and located at least 1,600 m from each other and permanent water sources. Each site was assigned a fall, spring, or non-burned fire treatment with 4 replicates for each of 2 years. Fire treatments were blocked by pasture to avoid potential effects of past or present grazing by cattle. Sand sagebrush was in an early stage of senescence during fall burns and leaves were 3 to 5 cm long at the time of spring burns. Plots supported 2,500 to 3,500 kg ha^{-1} of standing herbaceous fuel and good burning conditions allowed continuous fire fronts with nearly complete combustion of fine fuels and some 10-hr fuels (Table 4.1). Burned sites represented less than 2% of each pasture and were exposed to grazing by cattle from early April to September.

Prior to treatment, 20 live sand sagebrush plants were randomly selected in each plot, locations were mapped, and the shrubs were permanently marked by driving a red or white rebar stake measuring 38 cm at the base of each shrub. Shrub height and 2 horizontal canopy widths were measured to the nearest 1 cm before treatments were applied. Shrub height was measured from ground level to the highest live part. Canopy width was measured at its widest diameter and that perpendicular to the first, then used to calculate canopy area by the following formula:

$$\frac{\text{width 1} * \text{width 2} * \pi.}{4}$$

The product of canopy height and area was used to determine canopy volume.

All marked shrubs were monitored monthly from May through September in the year of treatment to assess survival and growth in canopy height, area, and volume. Canopy size as a percentage of pre-treatment measures was used in analyses of single-season shrub growth to reduce the influence of initial shrub size on subsequent regrowth. Plots burned in the first year were also examined in September of the second year to observe additional growth and possible delayed mortality.

The relationship between individual shrub size and mortality was assessed using point serial correlation (Lindman et al. 1980). The experimental units for the other analyses were the 4-ha plots. Canopy growth curves were described by regression analysis. All other data were analyzed by analysis of variance for a randomized block design (SAS Institute 1985). Sand sagebrush mortality was tested for differences by year and fire treatment. Canopy size was tested for differences by year, fire treatment, and month, with month as a repeated measure. Year was used as a repeated measure to test for differences in mortality and canopy growth by year and fire treatment for plots burned in the first year and monitored again the second year. A 5% significance level was used for all tests. When differences occurred and multiple comparisons were made, means were separated using Fisher's protected least significant difference (Steele and Torrie 1980).

Results

Sand sagebrush mortality was similar across treatment years (P>0.32) and varied among burn treatments (P<0.05). All non-burned shrubs survived during the study period, but mortality was about 4% on fall- and spring-burned plots. Sprouts originated primarily from the root crown and occasionally from buried lateral stems. Sprouting from old above-ground stems was rare and limited to shrubs that were not blackened by the fire. All of the marked shrubs were charred, or the above-ground portions were completely consumed by fire. Sprouting was positively correlated with shrub height (r=0.16, P<0.01), area (r=0.16, P<0.01), and canopy volume (r=0.15, P<0.01). Although correlation coefficients were small, some trends were observed in the relationship between shrub size and mortality. Prior to treatment, study plants ranged from 36 to 142 cm tall. No shrubs taller than 104 cm were killed by fire. When examined in 10-cm size classes, mortality was 3 to 4% (8/224) for classes between 70 and 109 cm tall, but 12% (5/43) for shrubs 60 to 69 cm tall and 21% (4/19) for those less than 60 cm tall.

There was a 3-way interaction between year, burn, and month for sand sagebrush height (P<0.01). The main effects of burn and month were significant (P<0.01), but year was not (P>0.30). Non-burned shrubs were consistently taller than burned shrubs, relative to pre-treatment measurements (Table 4.2). The relative size of burned plants was similar by August in 2000, but remained greater on fall burns than spring burns throughout the growing season in 2001. Canopy area was similar between years (P>0.65). However, there was a 2-way interaction between burn and month for canopy area (P<0.01), with all burn treatments being different through July (Table 4.3). Fall- and spring-burned shrubs were similar in relative area during August and September, remaining smaller than non-burned shrubs. Sand sagebrush canopy volume varied by month ($P<0.01$), but was similar between years ($P>0.66$). Each burn treatment differed from the others throughout the growing season ($P<0.01$). Non-burned shrubs were always largest and had positive growth relative to pre-treatment measurements. Averaged across months and years, canopy volume increased 16% for non-burned shrubs, but was reduced 66% by fall burns and 77% by spring burns.

Fire effects on sand sagebrush lasted through a second growing season and did not vary by season of burn for canopy height ($P>0.74$), area ($P>0.68$), or volume ($P>0.76$). Burned shrubs recovered to about 80% of their initial height and area, but canopy volume was still 38% lower after the second growing season (Table 4.4). Three additional shrubs died on burned plots after the first year, but delayed mortality was not a significant factor in sand sagebrush response to fire ($P>0.10$).

Sand sagebrush canopy growth was marginal and least predictable for non-burned shrubs (Fig. 4.1). Stem elongation began in May and most additional height was gained by July. Canopy area and volume continued to expand slowly through September. Fall and spring fires caused accelerated growth rates for sprouts and altered growth periods for the shrubs. The timing of growth for spring-burned shrubs was similar to that of non-burned shrubs, except canopy area and volume leveled off by August (Fig. 4.2). Nearly all stem growth on fall-burned shrubs occurred by July (Fig. 4.3). However, shrubs burned in fall initiated stem elongation in April, giving them a one-month advantage over other treatments.

Discussion and Conclusions

Burned sites were heavily grazed by cattle and growing seasons of both study years were dry, but these factors are believed to have had minimal effects on sand sagebrush growth and probably counteracted each other. Grass utilization by cattle was 81% on burned plots (Vermeire 2002), yet there were no signs of browsing on sand sagebrush sprouts and none of the study plants were trampled. No clear trends have been shown between grazing intensity and sand sagebrush cover (Sims et al. 1976, Collins et al. 1987), but intensive use of grasses would be expected to reduce competition with sand sagebrush. The shrubs also gave no visible indication of drought stress. Rasmussen and Brotherson (1984) showed sand sagebrush to be well-adapted to dry soils of low fertility. Collins et al. (1987) showed a positive correlation between sand sagebrush cover and precipitation, but the relationship was based on annual precipitation for the current and previous year. Although growing season precipitation was 10 and 25% below average in 2000 and 2001, annual precipitation was near the 62-year mean during both years of the study.

Most *Artemisia* shrub species in the United States are intolerant of fire (Dix 1960, Rowe 1969, Wright and Bailey 1982). Sand sagebrush is a strong exception to this tendency. The 96% sprouting frequency of sand sagebrush meets or exceeds that reported for mesquite (*Prosopis glandulosa* Torr.) and chamise (*Adenostoma fasciculatum* H. & A.), species known for their ability to sprout following fire (Wright et al. 1976, Keeley and Zedler 1978, Martin 1983). The positive correlation between shrub size and survival indicates that presumably younger plants are more susceptible to fire-induced mortality. This relationship has been documented for other woody species (Wright et al. 1976, Malanson and Trabaud 1988). Small correlation coefficients may have been a reflection of the population's age structure. Although it is considered rare for sand sagebrush to be taller than 1 m (Great Plains Flora Association 1986, Stubbendieck 1997), 21% of the shrubs exceeded this height. None of the marked shrubs were seedlings and the study area had not been burned or treated with herbicide for at least 15 years.

Sprouting has been described as an evolutionary means of existence in ecosystems prone to fire and other frequent disturbances (James 1984, Kruger et al. 1997). The contrast between fire effects on sand sagebrush and other *Artemisia* shrubs may be explained by differences in fire history. Prior to the 1900s, fire frequency in sagebrush grasslands west of the Rocky Mountains was 20 to 70 years (Houston 1973). Fires were small and infrequent in these shrublands because woody canopies reduced herbaceous production and disrupted the continuity of fine fuels (McLaughlin and Bowers 1982). However, the high productivity of sandy soils in the southern Great Plains and the volatility of sand sagebrush probably supported the same 5 to 10-year fire frequency experienced in Southern Plains grasslands. Sprouting ability of sand sagebrush in the Southwest and Great Basin is not documented and may vary from that in the Great Plains because of differences in fire history or precipitation.

Fire-induced reduction of sand sagebrush canopy lasted for at least 2 years and a third growing season would likely have been required for recovery to pre-treatment levels if growth rates remained constant. Although shrub density may be similar on burned and non-burned sites, it is conceivable that sand sagebrush cover would be lower with a 5 to 10-year fire return interval than is currently observed in sites deprived of fire. Prescribed fire may play a role in freshening sand sagebrush stands while maintaining shrub density. The survival and growth rates of burned sand sagebrush ensure that any loss of protective soil and wildlife cover is limited in duration. When density reduction is desired, alternative methods such as herbicide will be required. However, prescribed fire could probably extend the life of herbicide treatments by controlling some young plants.

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Table 4.1. Dates and average weather conditions for fall and spring prescribed burns conducted on Cooper Wildlife Management Area, Okla., 1999-2001.

Conditions	Burn			
	Fall		Spring	
Date	11/16/99	11/14/00	4/17/00	4/12/01
Air temperature (°C)	20	12	22	18
Relative humidity (%)	27	36	45	38
Wind speed (km hr^{-1})	6	12	10	8

Table 4.2. Sand sagebrush canopy height by year, burn, and month relative to pre-treatment measurements.

Month	2000			2001		
	Non-burned	Fall-burned	Spring-burned	Non-burned	Fall-burned	Spring-burned
	(%)					
May	100 a[1]	48 b	10 c	100 a	49 b	9 c
Jun.	106 a	70 b	56 c	103 a	73 b	48 c
Jul.	107 a	70 b	64 c	103 a	79 b	68 c
Aug.	106 a	68 b	65 b	104 a	79 b	67 c
Sep.	106 a	69 b	65 b	104 a	78 b	67 c

[1] Percentages followed by different letters within months and years are significantly different ($P<0.05$).

Table 4.3 Sand sagebrush canopy area by burn treatment and month relative to pre-treatment measurements.

Burn treatment	Month				
	May	Jun.	Jul.	Aug.	Sep.
	%				
Non-burned	100 a[1]	111 a	114 a	118 a	120 a
Fall-burned	20 b	47 b	57 b	57 b	58 b
Spring-burned	5 c	29 c	46 c	49 b	54 b

[1] Percentages followed by different letters within months are significantly different ($P<0.05$).

Table 4.4. Mean canopy height, area, and volume of sand sagebrush prior to prescribed fire and in September of the first and second years after burning.

Period	Height (cm)	Area (dm^2)	Volume (dm^3)
Pre-burn	82 a[1]	84.0 a	762.1 a
Year 1	56 b	44.1 b	273.7 b
Year 2	65 c	67.0 c	474.3 c

[1] Means followed by different letters within columns are significantly different ($P<0.05$).

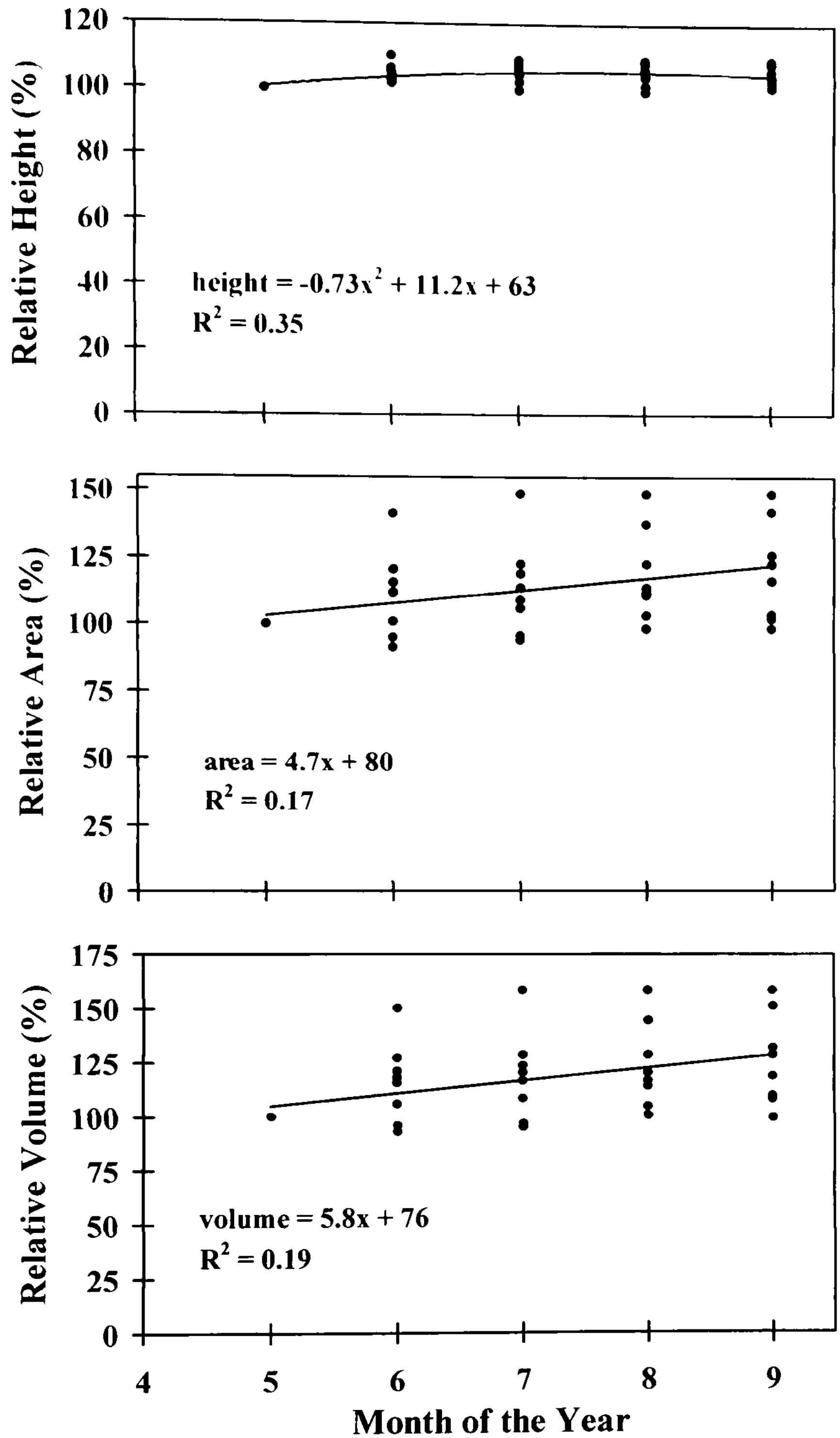

Figure 4.1. Changes in canopy height, area, and volume of non-burned sand sagebrush relative to pre-treatment measures.

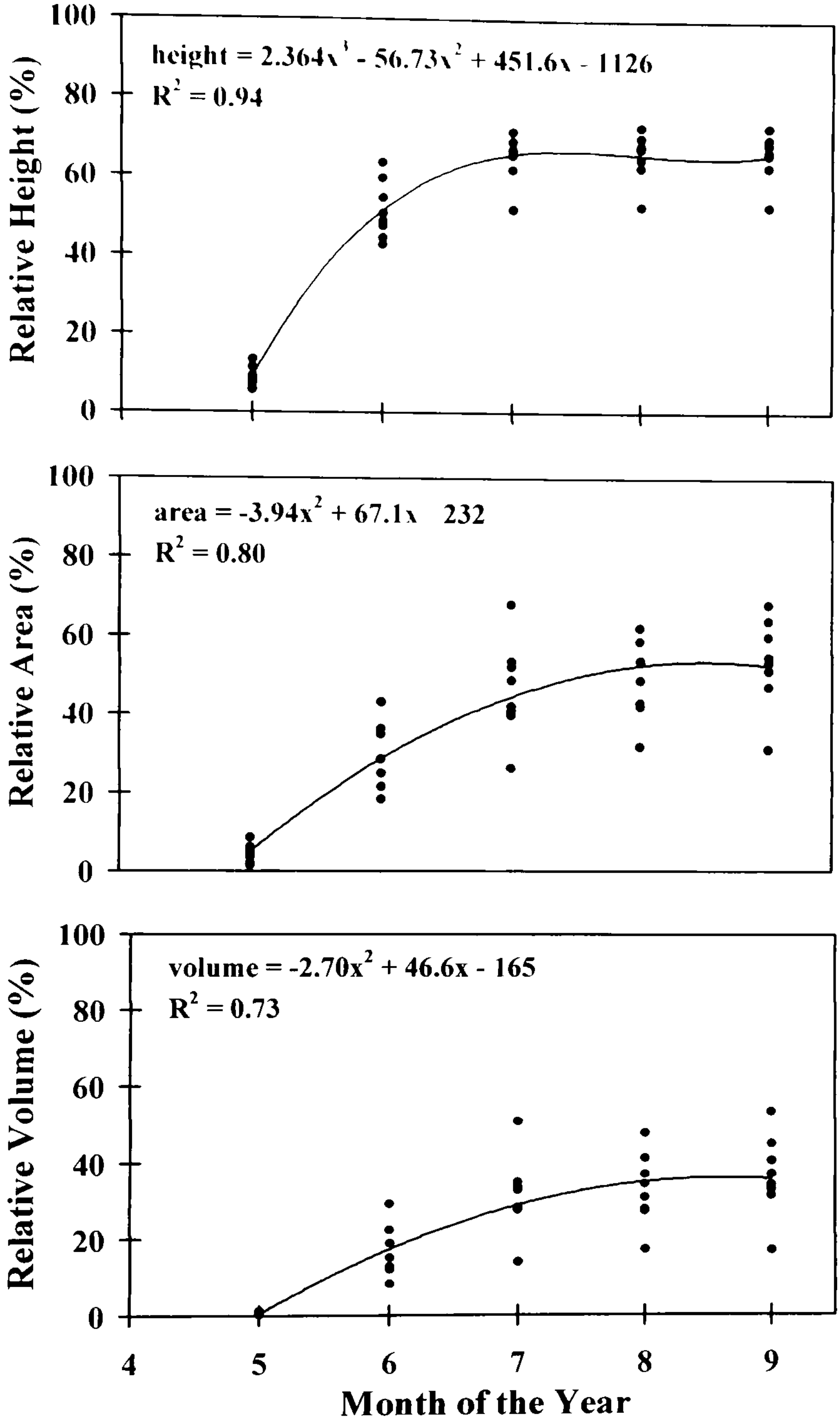

Figure 4.2. Changes in canopy height, area, and volume of spring-burned sand sagebrush relative to pre-treatment measures.

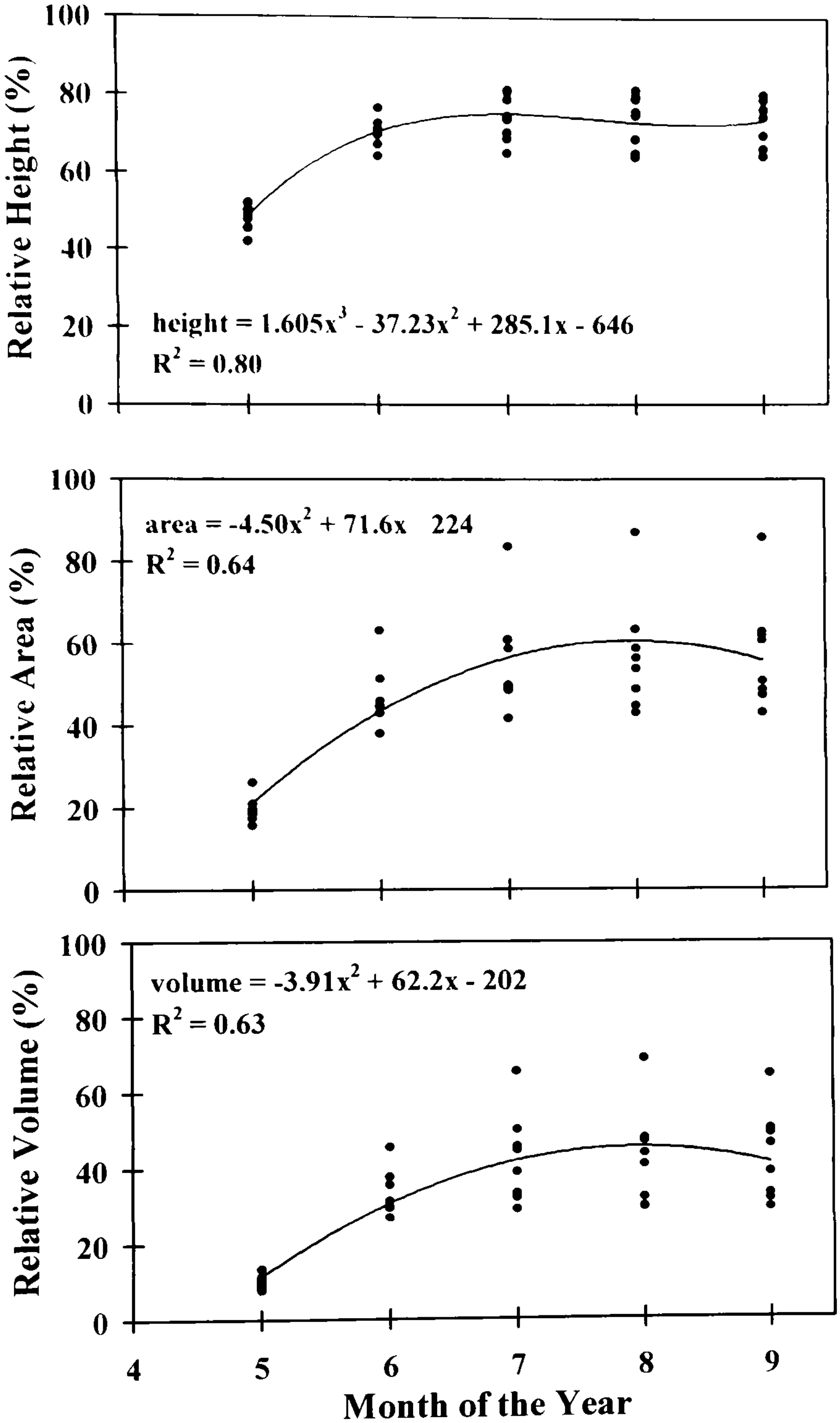

Figure 4.3. Changes in canopy height, area, and volume of fall-burned sand sagebrush relative to pre-treatment measures.

CHAPTER V

PATCH BURNING EFFECTS ON FORAGE UTILIZATION AND GRAZING DISTRIBUTION

Abstract

Post-fire forage growth is known to be a strong attractant for large herbivores. However, fire has generally been avoided as a grazing distribution tool for fear of localized over utilization of forage resources. Our objectives were to determine cattle grazing preference for burned sites relative to non-burned sites, examine whether forage utilization was affected by season of burn, determine forb response to patch burning, and describe the relationship between forage utilization and distance from burned sites. Sixteen 4-ha plots were burned in mid-November or mid-April and left exposed to cattle grazing for the duration of the growing season. Burn treatments were blocked within pastures to allow individual herds access to fall-burned, spring-burned, and non-burned sites. Standing crop estimates for grasses, forbs, and total herbage were made in September by clipping on burned sites and at 50, 100, 200, 400, and 800 m distant from the plot's edge. Standing crop was also sampled in exclosures on burned and non-burned sites. Cattle were strongly attracted to burned sites, utilizing 81% of the grasses within burns compared to 18% outside the influence of burns. Cattle showed no preference for one burn season over the other. Grass standing crop decreased in a predictable manner with proximity to burned plots. Grass utilization ranged from 59% at 50 m to 18% at 800

m away from burns. Forbs increased 60% on burned plots, but were unaffected by distance from burns. Patch burning can be employed as an effective, inexpensive grazing distribution tool.

Introduction

The art and science of rangeland management often revolve around the issue of animal selectivity and its effects on the distribution of their resource utilization. Generally, scientists and managers have sought uniform animal distribution to avoid having areas of over-utilized and under-utilized forage resources. However, forage utilization is infrequently uniform because combinations of biotic and abiotic characteristics are rarely homogeneous across the landscape and herbivores naturally have preferences for site conditions conducive to their needs. Understanding many of these preferences, herbivore distribution has been altered with strategic placement of attractants such as water (Valentine 1947, Martin and Ward 1970), shade (McIlvain and Shoop 1971), nitrogen fertilizer (Hooper et al. 1969, Samuel et al. 1980), salt, and supplemental feeds (Martin and Ward 1973, Bailey and Welling 1999). Fencing and implementing specialized grazing systems also make animal distribution more uniform by limiting choices available to the animals (Vallentine 1990). However, slope and distance to water are still overriding factors controlling distribution of forage use (Bailey et al. 1996).

Fire is a powerful tool that can alter animal distribution at various scales. Grazing distribution is often more uniform on burned pastures because differences in forage nutritive value, palatability, and accessibility among patches are reduced. Given the choice of burned or non-burned sites, large herbivores strongly select burned sites as long as forage quantity is adequate (Mitchell and Villalobos 1999). Recommendations have therefore been made to use prescribed burning only on a management-unit basis. Wright (1974) suggested burned patches should be protected by fencing, or the remainder of the unit should be burned to prevent heavy localized grazing and overuse.

Such a view has prevented fire from being used to its potential as a distribution tool. If fire effects on distribution of forage use are strong and predictable, patch burning could effectively be used to increase the uniformity of forage use, draw animals away from sensitive areas, or create greater landscape heterogeneity by encouraging concentrated forage use.

The objectives of this study were to determine cattle grazing preference for burned sites relative to non-burned sites, examine whether forage utilization was affected by season of burn, determine forb response to patch burning, and describe the relationship between forage utilization and distance from burned sites. We hypothesized that forage utilization would be greater on burned sites than non-burned sites, that forage utilization would be greater on spring-burned plots than fall-burned plots because of expected changes in forage production and species composition, that forb biomass would increase on and near burned sites, and that utilization would increase in a predictable manner with proximity to burned sites.

Methods and Materials

Study Area

The study was conducted in northwestern Oklahoma on the Hal and Fern Cooper Wildlife Management Area, about 15 km northwest of Woodward (36° 34' N, 99° 34' W, elev. 625 m). The area consists of sandhills vegetated by sand sagebrush (*Artemisia filifolia* Torr.) and high-seral mixed prairie. The mean annual precipitation is 572 mm, with about 70% occurring as rain during the April through September growing season. Mean monthly temperatures range from 1° C in January to 29° C in July (Unpublished data, Southern Plains Range Research Station). The area is lightly stocked with cow-calf and stocker herds at 22 AUD ha^{-1}, where grazing is initiated at the first of April and cattle are removed from the pastures in early September. Pastures are relatively large at 635 ha, but water is well-distributed throughout the study area, with most water sources within 3.2 km of another.

Data were collected on Deep Sand ecological sites with slopes of 1 to 12%. The dominant soils were Pratt loamy fine sands (sandy, mixed, mesic Lamellic Haplustalfs) and were interspersed with Tivoli fine sands (mixed, thermic Typic Ustipsamments) on the tops of dunes (Nance et al. 1960). Sand sagebrush was the dominant woody plant, providing 20 to 50% canopy cover over most of the area. Other woody plants included sand plum (*Prunus angustifolia* Marsh.), which occurred in isolated thickets, and eastern redcedar (*Juniperus virginiana* L.), which was sparsely distributed throughout the pastures. The herbaceous component was dominated by little bluestem [*Schizachyrium scoparium* (Michx.) Nash], gramas (*Bouteloua spp.* Lag.), western ragweed (*Ambrosia* *psilostachya* D.C.), sand bluestem (*Andropogon hallii* Hack.), sand lovegrass [*Eragrostis trichodes* (Nutt.) Wood], and Texas croton [*Croton texensis* (Kl.) Muell. Arg.].

Methods

I selected 16, 4-ha sites that were similar in vegetative composition and at least 1,600 m distant from each other and permanent water sources. Each site was assigned a fall or spring fire treatment so that 4 sites were burned in each season for each of 2 years. Fall burns were conducted on 16 November 1999 or 14 November 2000, when most warm-season plants were dormant. Spring burns were applied 17 April 2000 or 12 April 2001, when warm-season plants had only recently initiated growth. Burned sites represented less than 2% of each pasture and were exposed to grazing by cattle from early April to September. Burn treatments were blocked within pastures to allow individual cattle herds equal access to fall- and spring-burned plots. A cattle exclosure, measuring 5 x 10 m and constructed of wire panels with 10-cm mesh, was erected near the center of each burned plot and on 8 non-burned sites to allow estimates of forage availability.

Forage utilization was estimated in September by clipping end-of-season herbage standing crop. The sampling scheme consisted of a 100-m pace transect placed in the center of each burned plot and 50, 100, 200, 400, and 800 m from the edge of burns. Distances from burned plots were determined by following a compass and using a Yardage Pro 800 laser range finder (Bushnell Sports Optics, Overland Park, Kans.). At each distance, a 100-m transect, perpendicular to the line of travel, was paced and vegetation was sampled every 10 m. All forbs and grasses were clipped to ground level in 0.1 m^2 quadrats and bagged separately to determine grass, forb, and total herbage standing crop. Standing crop estimates were also determined from 10 quadrats within each of the cattle exclosures. Samples were air-dried to a constant weight at 53° C and weighed to the nearest 0.01 g.

Grass, forb, and total herbage standing crop were analyzed as a split block design using analysis of variance (SAS Institute 1985). The herbage components within burned plots were tested for differences by year, block, and season of burn. All other models included an additional term for distance from burned sites. A 5% significance level was used for all tests. When differences occurred and multiple comparisons were made, means were separated using Fisher's protected least significant difference (Steele and Torrie 1980).

Results and Discussion

Grass and total herbage standing crop were about 40% lower in 2001 than they were in 2000 ($P<0.01$, Fig. 5.1). Since forb standing crop was similar across years ($P>0.54$), changes in total herbage were primarily caused by reduced grass yields. Stocking rates were the same each year, so the sharp reduction in grass standing crop can probably be explained by differences in precipitation. Although annual precipitation was near the 62-year mean during both years of the study, growing season precipitation was 10% below the long-term average in 2000 and 25% below the long-term average in 2001.

Grass standing crop on burned plots was much lower than that across non-burned sites (P<0.01), averaging 365 kg ha^{-1} (Fig. 5.2). Only about 25% of the difference in standing crop could be attributed to standing dead material from previous years' growth, based on exclosure data. Standing crop estimates from exclosures indicated grass utilization was 81% on burned plots compared to 18% on non-burned sites 800 m away from burns. Although fire effects on the level of forage utilization have not been quantified previously, greater use was expected on burned sites because forage production, quality, and accessibility are commonly increased by fire (Wright and Bailey 1982) and the higher ratio of green *versus* senescent vegetation is believed to attract large herbivores (Stuth 1991, Mitchell and Villalobos 1999). Cattle were observed utilizing burned patches during their intensive morning and late afternoon feeding bouts. Herds remained near water sources during the warmer periods of the day. The increased grazing pressure on burned plots promoted forb production (P<0.01) and changed the sites from grass-dominated to forb-dominated communities. However, total herbage was less than half of that on non-burned plots (P<0.01) despite the 60% increase of forbs on burned plots (Fig. 5.2).

Cattle showed no preference for plots burned in one season over those burned in another, with grass standing crop on fall- and spring-burned plots being nearly identical (P>0.94, Fig. 5.3). Standing crop was similar between burn seasons for forbs (P>0.25) and total herbage (P>0.62) as well. The seasonal timing of fire generally affects plant species composition (Towne and Owensby 1984), which could be expected to alter use by herbivores. However, plant communities differed only slightly by season of burn

(Vermeire 2002). Unless the quantity of desirable forages is limited or foraging efficiency is reduced, forage quality will likely be the dominant factor in site selection by large herbivores. Bison (*Bison bison* L.) preference among tallgrass prairie sites burned in spring, summer, or fall was found to be minor despite measurable changes in plant community composition (Coppedge and Shaw 1998).

A positive quadratic relationship existed between grass standing crop and distance from burned plots ($P<0.01$), with 98% of the distance effect being explained by linear and quadratic terms (Fig. 5.4). Forage utilization ranged from 59% at 50 m to 18% at 800 m distant from burns. Most of the increased utilization occurred within 200 m, with grass standing crop increasing by about 6 kg ha^{-1} for each additional meter from the edge of burned patches. The increased forage use reported around dehydrated molasses was also focused within 200 m of the supplement, but utilization was relatively uniform in the area affected (Bailey and Welling 1999). The reduction in forage utilization with distance from burned sites was less gradual than has been shown for forage use around water sources in gentle terrain (Valentine 1947, Herbel et al. 1967, Martin and Cable 1974). Water and fire differ as distribution tools in that water is required, whereas burned sites are simply preferred. Forage quality is similar across distances from water prior to grazing-induced changes and cattle can camp near water as long as the forage supply is adequate. However, cattle must leave burned sites multiple times during the day unless water is available nearby. Given the contrast in forage quality on burned and non-burned sites and that water was located 1,600 m from the burns, there was little incentive for cattle to spend much time grazing non-burned vegetation surrounding burned sites.

Forb standing crop was similar across distances from burned plots at 674 kg ha^{-1} ($P>0.82$, Fig. 5.5). Although grass standing crop was reduced with proximity to burns, utilization levels were insufficient to promote a measurable forb response as was observed within burned plots. Grass utilization was less than 60% within the first 100 m and less than 40% between 200 and 800 m. Higher stocking rates would have increased forage utilization, but probably would not have altered the rate of change in grass or forb standing crop with distance from burns. These relationships were similar between years even with the growing-season drought of 2001 and the resulting increase in grazing pressure. Since forbs were unaffected by distance, the relationship between total herbage and distance from burns was similar to that of grass standing crop, differing only by the intercepts (Fig. 5.6).

Management Implications

Prescribed fire is among the most powerful grazing distribution tools available. We found cattle were willing to travel at least 1,600 m from water to utilize burned patches during their intensive feeding bouts. Since cattle showed no preference between sites burned in spring or fall, burn season could be selected to address other management goals with little or no effect on grazing use by cattle. Additionally, utilization of surrounding non-burned vegetation increased in a predictable manner with proximity to burned patches. These results indicate that grazing distribution can be controlled with some precision using prescribed fire. Burned patches could be strategically placed to attract cattle to underutilized portions of pastures, or to draw them away from sensitive areas, such as riparian zones. Fuhlendorf and Engle (2001) proposed that patch burning could also be used to increase heterogeneity across the landscape. The change from grass-dominated to forb-dominated communities on burned patches supported this hypothesis. Such changes in vegetative composition were limited to burned sites and would be expected to be short-lived, particularly if burned sites were traditionally avoided by livestock. In tallgrass prairie, grasses regained dominance within 2 or 3 years after patches were burned and grazed by bison (Coppedge et al. 1998). Our results indicate that prescribed fire is a powerful attractant for cattle and that it may provide an inexpensive, non-permanent alternative for manipulating grazing distribution.

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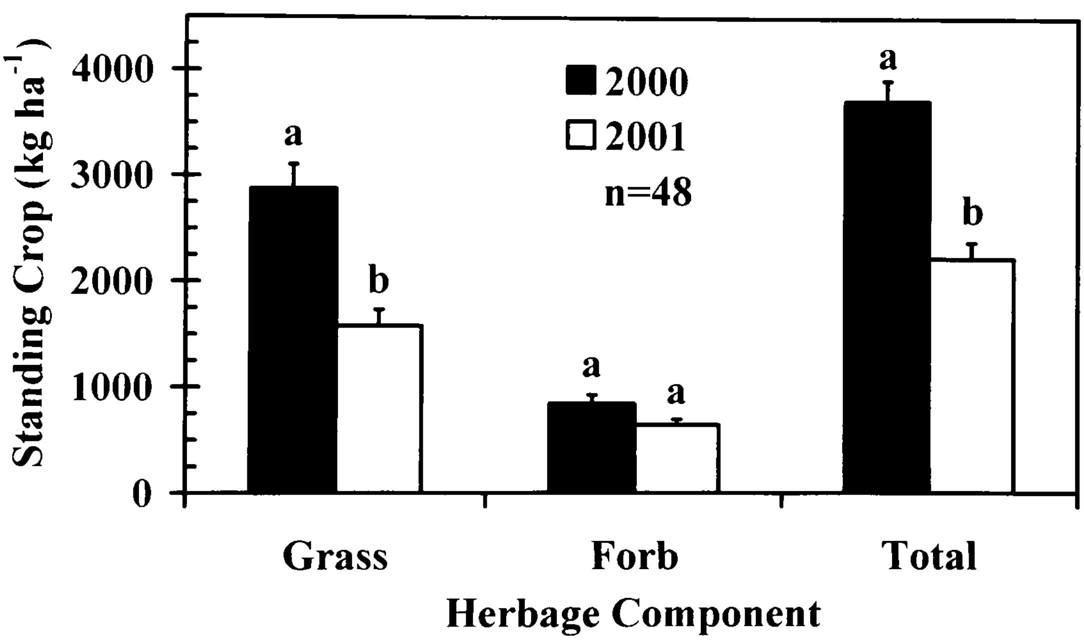

Figure 5.1. Standing crop of grasses, forbs, and total herbage across burn and distance treatments for 2000 and 2001. Means within herbage components with different letters are significantly different (P<0.05).

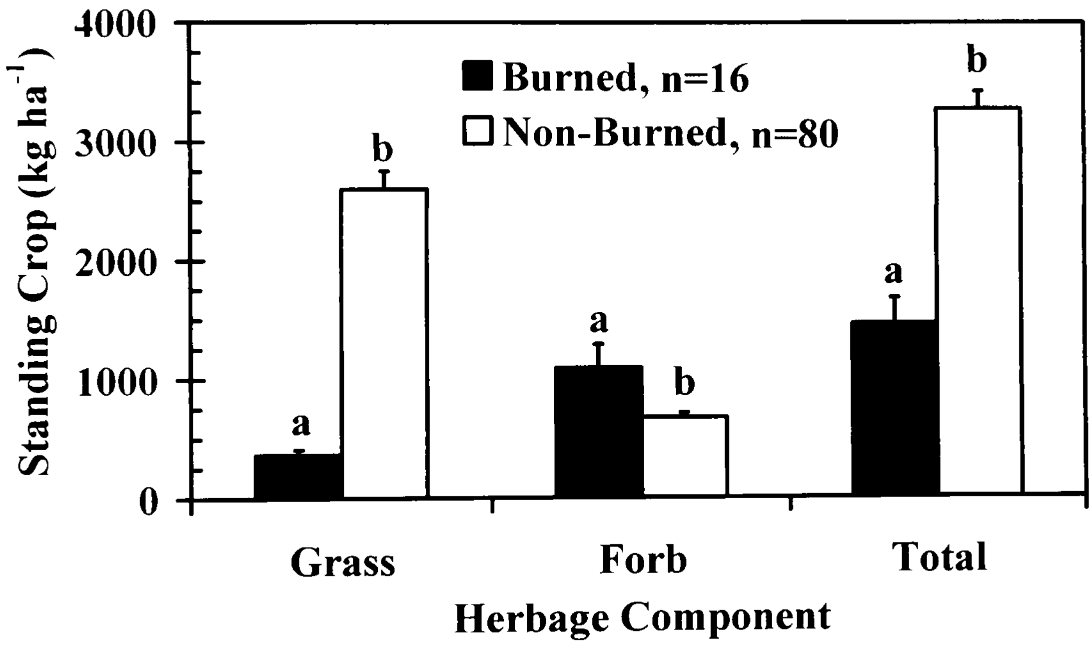

Figure 5.2. Standing crop of grasses, forbs, and total herbage across years for burned and non-burned sites. Means within herbage components with different letters are significantly different ($P<0.05$).

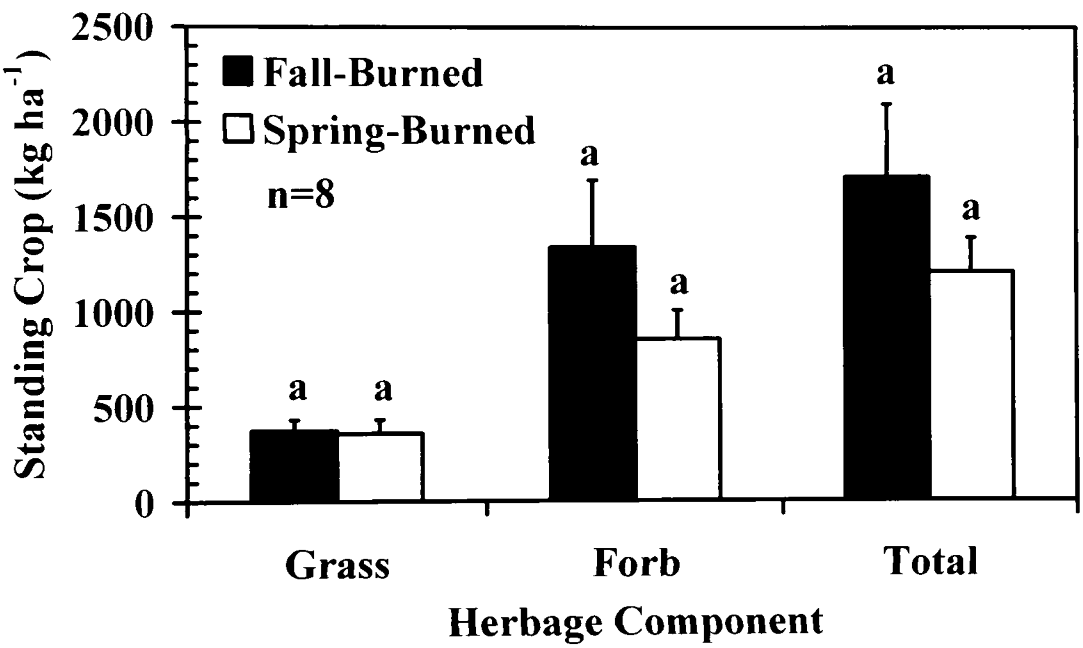

Figure 5.3. Standing crop of grasses, forbs, and total herbage across years and distance treatments for fall- and spring-burned plots. Means within herbage components with different letters are significantly different ($P<0.05$).

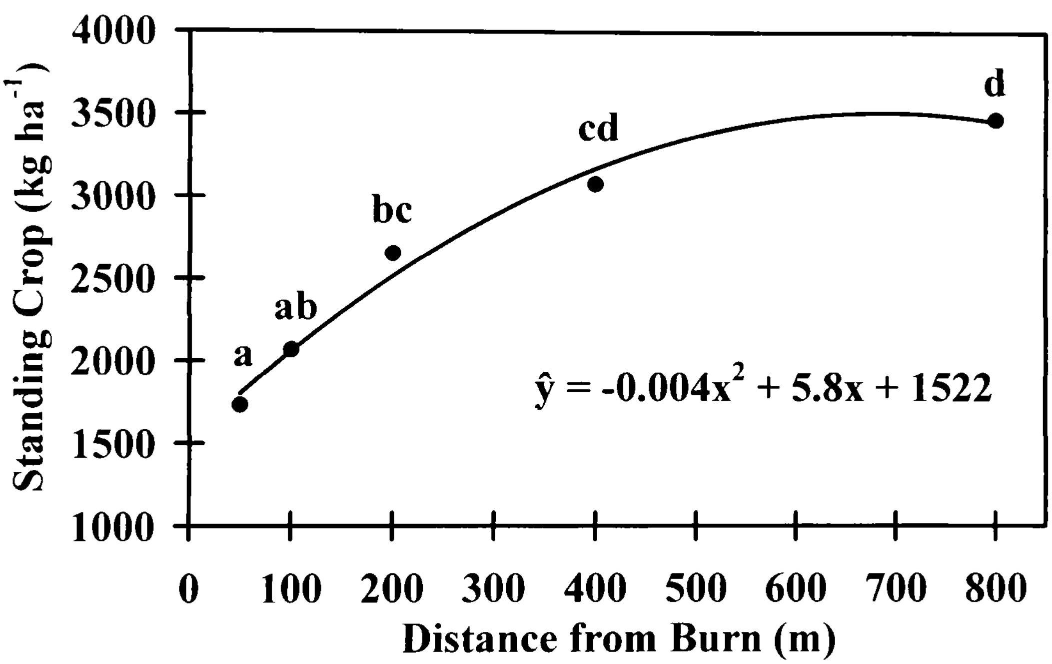

Figure 5.4. Relationship of grass standing crop with distance from burned patches across years and burn seasons. Distance means with different letters are significantly different ($P<0.05$).

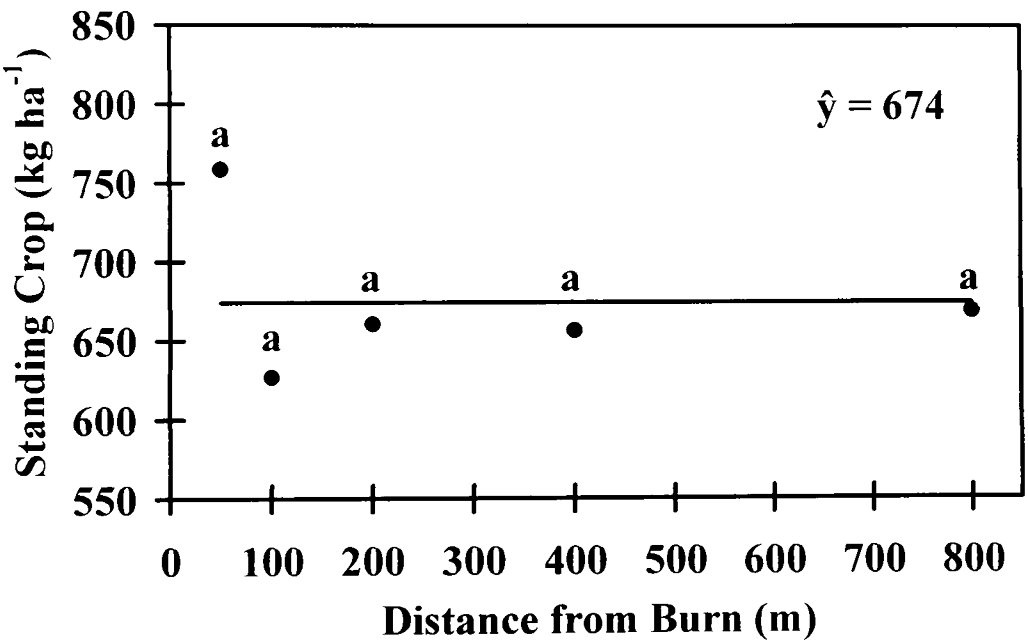

Figure 5.5. Forb standing crop at 5 distances from burned patches across years and burn seasons.

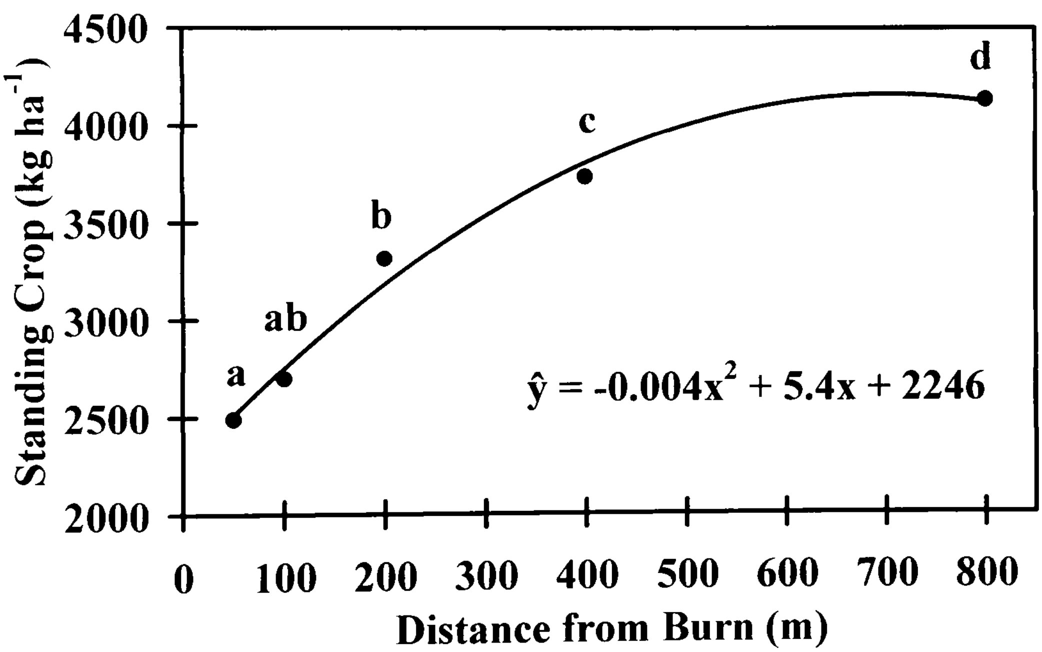

Figure 5.6. Relationship of total herbage standing crop with distance from burned patches across years and burn seasons. Distance means with different letters are significantly different ($P<0.05$).

CHAPTER VI

SELECTIVE CONTROL OF RANGELAND GRASSHOPPERS WITH PRESCRIBED FIRE

Abstract

Grasshoppers (Orthoptera: Acrididae) are considered among the most damaging rangeland pests, but are necessary for the survival of many wildlife species. Most grasshoppers are innocuous, but control with insecticides is non-discriminatory among species. The objectives were to evaluate the effects of prescribed burning on the abundance and biomass of grasshoppers and to determine if species could be selectively controlled with prescribed fire. Twenty-four 4-ha sites were selected in a sand sagebrush-mixed prairie near Woodward, Okla. and blocked by pasture. Plots were randomly assigned fall-, spring-, or non-burned treatments within block with 4 replications per treatment for each of 2 years. Grasshopper biomass and abundance were sampled in late July and early August by sweeping with canvas beating nets. Specimens were weighed to the nearest 0.1 mg and identified to species. Fire treatments had no effects on the total abundance or biomass of grasshoppers across species, with about 10 grasshoppers weighing 4,090 mg per 150 sweeps. Fire effects on the 4 most common species were variable and could be explained by the biology of the insects. *Melanoplus bowditchi* and *M. flavidus* were unaffected by fire treatment. *Hesperotettix viridis* is sensitive to damage to its host plants and was reduced about 88% by fire in either season.

Fall burns reduced *Ageneotettix deorum* abundance by 65% because the species' eggs are laid near the soil surface and exposed to the heat of passing fire. Fire prescriptions can be written to target species-specific vulnerabilities and control pest grasshoppers while maintaining the food base for grasshopper predators.

Introduction

Grasshoppers (Orthoptera: Acrididae) are considered among the most detrimental invertebrate pests throughout the world because of the damage they inflict on agricultural crops and forage resources (Watts et al. 1989). Forage consumption rates of individual grasshoppers are relatively small, but they waste nearly as much as they consume and at high densities, can substantially reduce forage availability for livestock and herbivorous wildlife. Hewitt and Onsager (1983) estimated that more than 20% of the forage in the western United States is lost to grasshoppers annually.

Control efforts with insecticides have been controversial because of the costs, short treatment life, and potential effects on non-target species (Blickenstaff et al. 1974, Watts et al. 1989). Although grasshoppers compete with other herbivores and some species are agricultural pests, they also provide an important seasonal food source for many wildlife species, particularly birds. Insecticides used for grasshopper control have had few direct effects on birds (McEwen et al. 1972, Stromborg et al. 1984, George et al. 1995). However, with grasshopper crude protein concentrations of 50 to 70% (Ueckert et al. 1972, DeFoliart 1975), predators would have difficulty overcoming their absence following large-scale control. Grasshoppers are a large component of summer and fall diets for gallinaceous birds (Davis et al. 1975, Doerr and Guthery 1983) and along with other insects, have been considered necessary for the survival and growth of chicks (Johnson and Boyce 1990).

Of the 600 grasshopper species in the United States, only about a dozen frequently occur at high densities. Selective control of these destructive species would maintain the food base for insectivores while protecting the forage base for herbivores. Prescribed fire is a potential alternative to chemical control that may allow more specific targeting of pest species. Direct mortality is likely if burns are conducted when grasshoppers are in a wingless nymph stage, or if eggs are exposed to lethal temperatures. Variations in grasshopper hatching dates and methods of egg deposition may allow fire prescriptions to be directed at species-specific control. Our objectives were to evaluate the effects of fall and spring prescribed burning on the abundance and biomass of grasshoppers and to determine if species could be selectively controlled with prescribed fire.

Methods and Materials

Study Area

The study was conducted about 15 km northwest of Woodward, Okla. (36° 34' N, 99° 34' W, elev. 625 m) on the Hal and Fern Cooper Wildlife Management Area. The site is a mixed prairie shrubland on undulating sandhills with slopes of 1 to 12%. Mean annual precipitation is 572 mm with 70% occurring as rain during the growing season

(Apr-Sep). Mean monthly temperatures range from 1° C in January to 29° C in July (Unpublished data, USDA-ARS).

All data were collected on Deep Sand ecological sites. The dominant soils are Pratt loamy fine sands (sandy, mixed, mesic Lamellic Haplustalfs) except on the tops of dunes where Tivoli fine sands (mixed, thermic Typic Ustipsamments) are common (Nance et al. 1960). Sand sagebrush (*Artemisia filifolia* Torr.) was the principal woody plant with 20 to 50% canopy cover across most of the area. Isolated thickets of sand plum (*Prunus angustifolia* Marsh.) were also present. The herbaceous component was dominated by little bluestem [*Schizachyrium scoparium* (Michx.) Nash], gramas (*Bouteloua spp.* Lag.), western ragweed (*Ambrosia psilostachya* D.C.), sand bluestem (*Andropogon hallii* Hack.), sand lovegrass [*Eragrostis trichodes* (Nutt.) Wood], and Texas croton [*Croton texensis* (Kl.) Muell. Arg.]. Pre-treatment herbaceous standing crop was 2,800 to 3,500 kg ha^{-1}. Pastures were lightly stocked (21-23 AUD ha^{-1}) with cattle annually from April to September.

Methods

We selected 24, 4-ha plots with the restrictions that they were similar in vegetative composition and located at least 1.6 km apart. Plots were blocked by pasture and randomly assigned fall-burned, spring-burned, or non-burned treatments within blocks, with 4 replications per burn treatment for each of 2 years. Fall burns were conducted on 16 November 1999 and 14 November 2000. Spring burns were applied 17

April 2000 and 12 April 2001. No restrictions were placed on cattle grazing within pastures during the grazing season.

Grasshoppers were sampled between 1100 and 1600 h with light (8-19 km h^{-1}) winds and hot (32-38° C), clear (< 10% clouds) weather conditions at the end of July 2000 and in mid-August 2001. Sampling was conducted by sweeping a standard canvas beating net (38-cm diameter) through the top layer of vegetation, making a 180° arc at each step. Each plot was sampled with 150 sweeps from 3 randomly located 50-sweep transects. After each set of 50 sweeps, net contents were dumped into 3.8-L plastic bags that were sealed then placed in a dark plastic bag. Specimens were placed on ice after each plot was sampled then frozen until sorted in the lab. Adult and late-instar grasshoppers were counted, weighed to the nearest 0.1 mg, and identified to species using keys (Helfer 1953, Coppock 1962, Otte 1981, 1984, Pfadt 1988).

Weather and cattle grazing were monitored throughout the study since both factors have been shown to affect grasshopper populations (Fielding and Brusven 1990, Onsager 2000). Temperature and precipitation data were obtained from a local weather station on the Southern Plains Experimental Range 10 km from the study site. Mean daily air temperatures from April through September were converted to degree-days above 17.8° C, based on a threshold for nymphal development (Putnam 1963).

Data were analyzed as a randomized block design with analysis-of-variance (SAS Institute 1985). Models for grasshopper abundance, biomass, and abundance of common individual species or species complex included terms for year, block within year, burn treatment, and the interactions. When differences occurred, means were separated using

Fisher's Protected Least Significant Difference (Steele and Torrie 1980). An alpha level of 0.10 was used for all tests.

Results and Discussion

Weather conditions were favorable for grasshoppers both years of the study, with mean growing- and dormant-season air temperatures of about 23 and 6° C. The period from April through September provided ample opportunity for nymphal development with 1,058 and 1,091 degree-days above 17.8° C in 2000 and 2001, respectively. Annual precipitation was near normal, but growing-season precipitation was 9 and 26% below the 62-year mean for 2000 and 2001 (Fig. 6.1). Grass utilization by cattle was less than 20% on non-burned plots, but cattle were strongly attracted to plots burned in spring or fall, utilizing more than 80% of the grasses (Vermeire 2002).

Grasshopper abundance across species was greater in 2000 than 2001 at 13 and 8 grasshoppers per 150 sweeps ($P<0.03$), but was not affected by burn treatment ($P>0.38$). These results were contrary to expectations, given the conditions believed to favor grasshopper populations. Grasshopper abundance in the southern Great Plains has been reported to be strongly favored by dry years and heavy grazing (Smith 1940, Campbell et al. 1974). However, the lower abundance during the 2001 drought and lack of burn treatment differences indicate total grasshopper abundance was unaffected by grazing pressure and may have been reduced by the drought. Despite differences in abundance, grasshopper biomass was similar between years ($P>0.10$) and among burn treatments ($P>0.46$) at 4,092 mg per 150 sweeps.

Fifteen species of grassoppers were collected with 4 accounting for 83% of the individuals (Table 6.1). The most common species were *Melanoplus bowditchi* and *M. flavidus*, representing 61% of the grasshoppers captured. Both species were present based on males collected, but were classified as a complex because of their similar habits and the inability to distinguish between females. *Ageneotettix deorum* and *Hesperotettix viridis* were the next largest groups, representing 16 and 6% of the collection, respectively. *Melanoplus bowditchi*, *M. flavidus*, and *H. viridis* are forbivorous and generally considered innocuous species, but the graminivorous *A. deorum* has been ranked the fifth most detrimental grasshopper on Western rangelands (Dysart 1995).

Prescribed fire did not affect abundance of the *M. bowditchi/flavidus* complex ($P>0.81$, Fig. 6.2). Given the reproductive ecology of these species and the timing of the burns, direct mortality was unlikely. Eggs are insulated from fire by being laid deeply in the soil and nymphs do not hatch until late April or early May (Pfadt 1988). The survival and regrowth of sand sagebrush on burned plots contributed to the fire tolerance of *M. bowditchi* as well. *Melanoplus bowditchi* is known to forage only on plants in the genus *Artemisia* and the majority of their time is spent on sagebrush.

Hesperotettix viridis abundance was reduced 92 and 85% by fall and spring prescribed fires, respectively ($P<0.03$, Fig. 6.2). However, we do not believe the decreases were direct effects of fire in either season. Hatching dates and egg deposition of *H. viridis* are similar to those of *M. bowditchi* and *M. flavidus* (Pfadt 1988), leaving eggs and nymphs unexposed to our fires. Fire-induced changes in plant species composition should have favored *H. viridis*. Western ragweed was common across treatments and is a known host plant for *H. viridis*, but the species utilizes numerous plants in the Asteraceae family that were available. The drastic reductions in *H. viridis* abundance may have been in response to the physical damage to host plants by fire or subsequent grazing activities by cattle. *Hesperotettix viridis* has been shown to avoid plants damaged by herbivory or water stress and populations have crashed following their own intensive herbivory on host plants (Parker 1984).

The abundance of *A. deorum* was 65% lower on fall-burned plots than non-burned plots ($P<0.10$, Fig. 6.2). Mean abundance on spring-burned plots was 40% lower than non-burned plots, but did not differ statistically from fall- or non-burned plots ($P>0.10$). Fall burns may have reduced populations through a number of direct and indirect factors related to the reproductive ecology of *A. deorum*. *Ageneotettix deorum* lays its eggs horizontally near the soil surface (Pfadt 1988), leaving them exposed to the heat of passing fire. Those not killed by the high temperatures would have been susceptible to erosion of the surrounding soil, more vulnerable to predation, and less insulated from weather for about 4 months before they could hatch safely. Spring burns should have had equal or greater potential for direct mortality than fall burns since some of the *A. deorum* eggs generally hatch by mid-April in the southern Great Plains (Pfadt 1988). Burning at this time would kill any of the wingless nymphs and potentially damage the remaining eggs. *Ageneotettix deorum* abundance on 1 spring-burned plot was 10 times greater than that of the other 7 plots. Areas adjacent to this plot held standing water for a period in late May and early June. Otherwise, the cause for this difference is unknown. With the exclusion of this plot, *A. deorum* abundance was reduced 71% by spring fires.

Conclusions

Fire is a natural phenomenon that has been shown to affect grasshopper assemblages, presumably through alterations in vegetative structure and species composition (Evans 1984, 1988). However, the influence of fire on grasshoppers is not limited to indirect effects of altering habitat. Fires can be prescribed to selectively control some grasshoppers based on unique biological characteristics. Because of the egg deposition habits of *Ageneotettix deorum*, we were able to control one of the most detrimental grasshoppers in the western United States without reducing the overall abundance or biomass of grasshoppers. Many of the pest grasshopper species have traits that should make them vulnerable to fire. Species that overwinter as nymphs, such as *Eritettix simplex* (Scudder) and *Xanthippus corallipes* (Haldeman), could be specifically targeted by winter burns because the nymphs are relatively immobile and other species are protected in the soil as eggs. *Aulocara elliotti* (Thomas), ranked the second most detrimental grasshopper by Dysart (1996), lays its eggs near the soil surface like *Ageneotettix deorum*. The ability to kill the eggs of such species will likely depend on heat intensity and duration. Both of these factors are maximized if fuels are dry, abundant, and continuous at the time of burning. Possibilities also exist for indirect control of early-hatching species. Soils on burned sites typically warm earlier in the spring than non-burned sites (Wright and Bailey 1982), so fire may encourage an early hatch and expose nymphs to late-winter and early-spring freezes.

The goals of grasshopper control programs are generally to reduce the frequency, duration, or magnitude of outbreaks by pest species. Prescribed fire may be a satisfactory alternative to insecticides that would accomplish these goals while maintaining the food base for grasshopper predators, such as gallinaceous birds. Additional research is needed to explore the possibility of prescribed fire as a control agent for pest species in other regions.

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Table 6.1. Relative abundance and pest-status ranking of grasshopper species collected near Woodward, Okla. in 2000 and 2001.

Species	Relative abundance (%)	Pest-status rank [1]
Melanoplus bowditchi / flavidus (Scudder) [2]	60.6	82, 70
Ageneotettix deorum (Scudder)	15.8	5
Hesperotettix viridis (Scudder)	6.4	377
Mermiria bivittata (Serville)	5.6	17
Hippiscus ocelote (Saussure)	3.6	69
Spharagemon cristatum (Scudder)	2.4	254
Arphia xanthoptera (Burmeister)	1.6	129
Melanoplus bivittatus (Say)	1.2	4
Psinidia amplicornis (Caudell)	0.8	360
Arphia simplex (Scudder)	0.4	128
Chortophaga viridifasciata (DeGeer)	0.4	58
Melanoplus sanguinipes (Fabricius)	0.4	1
Opeia obscura (Thomas)	0.4	14
Schistocerca obscura (Fabricius)	0.4	159

[1] Based on the ranking of 377 species in a literature review by Dysart (1996).

[2] *M. bowditchi* and *M. flavidus* were classified as a complex because females were indistinguishable.

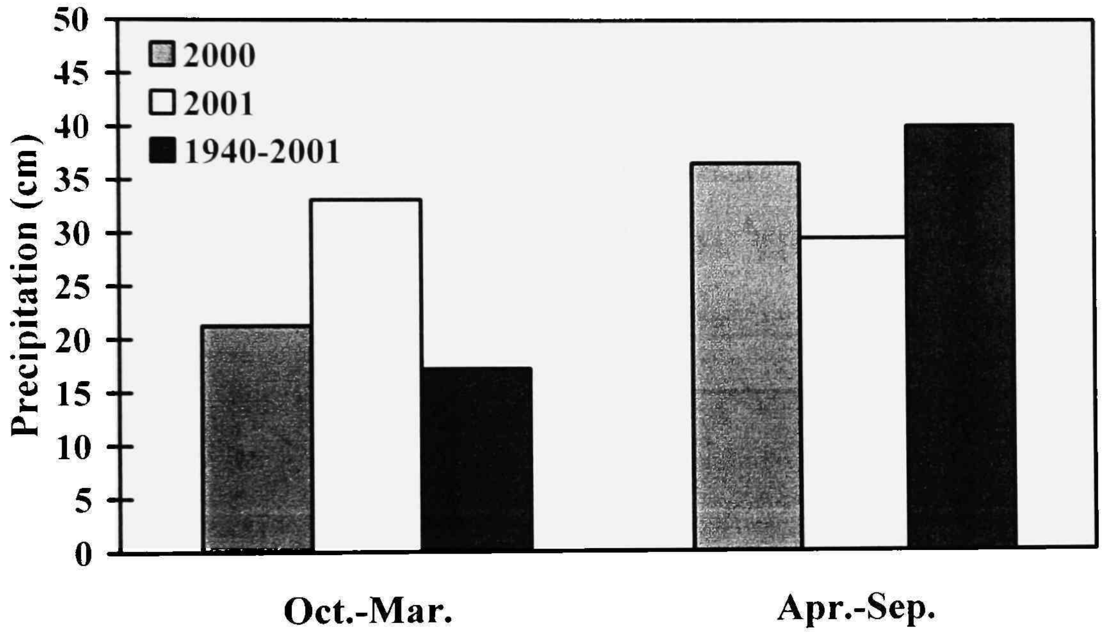

Figure 6.1. Dormant (Oct.-Mar.) and growing-season (Apr.-Sep.) precipitation for Woodward, Okla. in 2000, 2001, and the 62-year mean from 1940 through 2001.

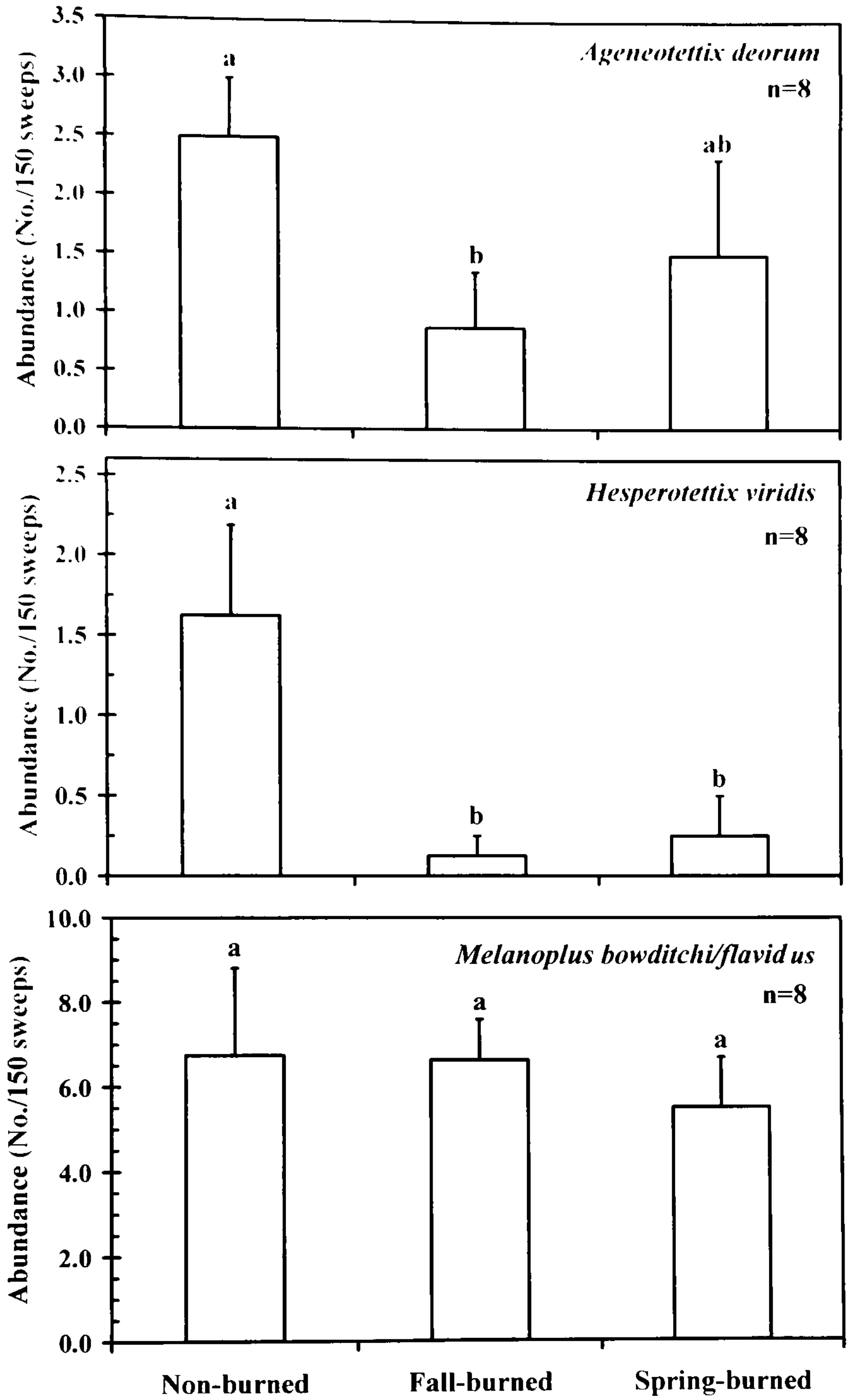

Figure 6.2. Prescribed fire effects on the abundance of *Ageneotettix deorum*, *Hesperotettix viridis*, and a *Melanoplus bowditchi/flavidus* complex on non-burned, fall-burned, and spring-burned sites in sand sagebrush-mixed prairie.

CHAPTER VII

CONCLUSIONS

The physiology and sprouting ability of sand sagebrush make this shrub well-adapted to fire and other disturbances. This may be expected since sand sagebrush has evolved on sandy soils susceptible to regular shifting by wind and was likely exposed to frequent fires in the Great Plains. The limited period of physiological vulnerability to disturbance and the capacity to sprout new stems indicate sand sagebrush has long played a prominent role in the structure and function of the sandy rangelands where it occurs, although the size and density of the shrubs may have been less than what is currently observed on fire-deprived landscapes.

The total non-structural carbohydrate trends and sprouting ability of sand sagebrush offer little opportunity for control. Fire or mechanical treatments should have little impact on sand sagebrush unless applied repeatedly during stem elongation. Treatments applied to the canopy from maturity through dormancy should allow removal of decadent shrub material with little or no damage to sand sagebrush stands. Although periods of rapid canopy growth have been recommended for treating sand sagebrush with foliar-applied herbicides, our results indicate herbicides should be applied when stem growth has stalled and flowering has been initiated, unless the cuticle is not receptive during this period. Coordinating management with total non-structural carbohydrate trends should promote desired results with reduced effort and costs.

Most *Artemisia* shrub species in the United States are intolerant of fire. Sand sagebrush is a strong exception to this tendency. The contrast in fire effects on sand sagebrush and other *Artemisia* shrubs may be explained by differences in fire history since fires were historically more frequent in the Great Plains than sagebrush grasslands farther west. Sprouting ability of sand sagebrush has not been determined at the western extent of the shrub's distribution in the Southwest and Great Basin, but may differ because of variation in fire history.

Prescribed fire may play a role in freshening sand sagebrush stands while maintaining shrub density. The survival and growth rates of burned sand sagebrush ensure that any loss of protective soil and wildlife cover is limited in duration. Survival rates are high, but fire can significantly reduce sand sagebrush canopies for at least 2 or 3 years. If density reduction is desired, alternative methods such as herbicide will be required. However, the correlation between plant size and mortality indicates prescribed fire could probably extend the life of herbicide treatments by controlling some young plants.

Prescribed fire is among the most powerful grazing distribution tools available. Cattle were willing to travel at least 1,600 m from water to utilize burned patches during their intensive feeding bouts. Since cattle showed no preference between sites burned in spring or fall, burn season could be selected to address other management goals with little or no effect on grazing use by cattle. Utilization of surrounding non-burned vegetation increased in a predictable manner with proximity to burned patches, indicating that grazing distribution can be controlled with some precision using prescribed fire. Burned patches could be strategically placed to attract cattle to underutilized portions of pastures, or to draw them away from sensitive areas, such as riparian zones. Alternatively, patch burning could be used to increase landscape heterogeneity by creating islands of forb-dominated communities. Such changes in vegetative composition were limited to burned sites and would be expected to be short-lived, particularly if burned sites were traditionally avoided by livestock. Our results indicate that prescribed fire is a powerful attractant for cattle and may provide an inexpensive, non-permanent alternative for manipulating grazing distribution and subsequent changes in the structure and composition of plant communities.

Prescribed fire may also provide an alternative to broad-scale control of grasshoppers with pesticides. Some grasshopper species have unique biological characteristics that make them vulnerable to fire. We were able to control *Ageneotettix deorum*, one of the most detrimental grasshoppers in the western United States, without reducing the overall abundance or biomass of grasshoppers because of its egg deposition habits. Many of the pest grasshopper species have similar traits that should make them susceptible to fire, such as shallow egg deposition and overwintering as nymphs. Therefore, it appears fires can be prescribed to selectively control some grasshoppers. The goals of grasshopper control programs are generally to reduce the frequency, duration, or magnitude of outbreaks by pest species. Prescribed fire may be a satisfactory alternative to insecticides that would accomplish these goals while maintaining the food base for grasshopper predators.

APPENDIX

DATA

Table A.1. Mean root total non-structural carbohydrate (TNC) concentrations for sand sagebrush, mean daily temperature, and precipitation by month and year on the Hal and Fern Cooper Wildlife Management Area, Fort Supply, OK.

Month	Year	Mean daily temperature	Precipitation	Root TNC
		--- (°C) ---	--- (cm) ---	--- (%) ---
Nov	1999	12.8	0.0	36.85
Dec	1999	4.2	1.1	32.31
Jan	2000	3.8	0.0	33.15
Feb	2000	2.7	1.9	34.90
Mar	2000	8.7	15.3	28.68
Apr	2000	13.1	2.5	31.42
May	2000	17.6	2.9	25.80
Jun	2000	22.3	26.0	18.89
Jul	2000	26.2	4.9	34.28
Aug	2000	27.5	1.6	.
Sep	2000	29.0	0.1	44.94
Oct	2000	20.9	3.3	48.94
Nov	2000	11.6	13.8	47.84
Apr	2001	10.7	5.0	16.57
May	2001	17.8	9.5	.
Jun	2001	21.0	5.6	14.03
Jul	2001	27.0	3.1	14.85
Aug	2001	29.5	1.7	19.17
Sep	2001	24.6	3.5	26.65

Table A.2. Sand sagebrush canopy height, area, and volume by block, year, and month for non-burned (0), fall-burned (1), and spring-burned (2) shrubs on the Hal and Fern Cooper Wildlife Management Area, Fort Supply, OK in 2000 and 2001.

Block	Year	Burn	Month	Height (cm)	Area (dm^2)	Volume (dm^3)
1	1	0	0	76	99.1	909.4
2	1	0	0	76	82.5	723.9
3	1	0	0	69	78.5	660.1
4	1	0	0	71	88.0	694.3
1	1	1	0	79	74.3	617.7
2	1	1	0	85	75.9	675.1
3	1	1	0	83	96.1	922.7
4	1	1	0	82	84.6	758.4
1	1	2	0	67	54.2	384.2
2	1	2	0	88	106.1	1054.1
3	1	2	0	88	82.3	777.0
4	1	2	0	88	97.9	927.6
1	1	0	5	76	99.1	909.4
2	1	0	5	76	82.5	723.9
3	1	0	5	69	78.5	660.1
4	1	0	5	71	88.0	694.3
1	1	1	5	41	15.7	70.1
2	1	1	5	36	13.1	55.4
3	1	1	5	40	18.4	87.6
4	1	1	5	41	16.7	76.7
1	1	2	5	9	3.3	3.2
2	1	2	5	10	9.1	12.2
3	1	2	5	6	1.7	1.5
4	1	2	5	5	1.5	1.2
1	1	0	6	81	140.0	1363.0
2	1	0	6	83	99.3	917.7
3	1	0	6	73	87.6	761.2
4	1	0	6	74	98.5	819.9
1	1	1	6	55	33.1	192.5
2	1	1	6	55	34.0	199.7
3	1	1	6	59	41.3	270.9
4	1	1	6	60	43.5	284.2
1	1	2	6	43	23.4	112.0
2	1	2	6	53	38.6	233.9
3	1	2	6	48	28.8	145.6
4	1	2	6	38	18.0	77.2
1	1	0	7	82	147.4	1439.0

Table A.2. Continued.

Block	Year	Burn	Month	Height (cm)	Area (dm^2)	Volume (dm^3)
2	1	0	7	83	101.1	927.6
3	1	0	7	73	89.3	791.4
4	1	0	7	75	96.1	806.8
1	1	1	7	54	31.0	177.9
2	1	1	7	55	36.8	215.0
3	1	1	7	58	47.4	305.8
4	1	1	7	62	51.4	345.4
1	1	2	7	48	36.9	194.8
2	1	2	7	61	51.6	353.9
3	1	2	7	54	43.9	253.4
4	1	2	7	46	25.7	128.9
1	1	0	8	82	147.4	1439.0
2	1	0	8	83	101.1	927.6
3	1	0	8	73	89.3	791.4
4	1	0	8	72	97.7	790.2
1	1	1	8	51	32.0	182.6
2	1	1	8	55	34.0	198.6
3	1	1	8	57	47.0	294.7
4	1	1	8	62	53.8	355.9
1	1	2	8	49	33.6	182.8
2	1	2	8	62	62.1	432.2
3	1	2	8	55	44.0	239.5
4	1	2	8	46	31.1	159.1
1	1	0	9	82	147.4	1439.0
2	1	0	9	83	101.1	927.6
3	1	0	9	72	91.2	781.1
4	1	0	9	73	91.4	758.3
1	1	1	9	51	32.0	180.0
2	1	1	9	56	36.9	222.4
3	1	1	9	58	45.5	290.3
4	1	1	9	61	52.4	348.7
1	1	2	9	49	36.9	204.8
2	1	2	9	62	67.8	471.7
3	1	2	9	55	45.0	263.7
4	1	2	9	46	30.5	156.2
1	2	0	0	87	97.1	967.3
2	2	0	0	82	70.7	599.8
3	2	0	0	89	101.7	979.9
4	2	0	0	79	70.5	574.6
1	2	1	0	90	131.0	1294.9

Table A.2. Continued.

Block	Year	Burn	Month	Height (cm)	Area (dm^2)	Volume (dm^3)
2	2	1	0	91	124.9	1212.6
3	2	1	0	92	118.2	1132.5
4	2	1	0	104	115.0	1260.3
1	2	2	0	91	88.8	849.4
2	2	2	0	90	100.8	945.5
3	2	2	0	87	104.7	1008.4
4	2	2	0	88	95.7	923.4
1	2	0	5	87	97.1	967.3
2	2	0	5	82	70.7	599.8
3	2	0	5	89	101.7	979.9
4	2	0	5	79	70.5	574.6
1	2	1	5	44	20.6	101.5
2	2	1	5	42	23.0	110.2
3	2	1	5	48	22.5	118.1
4	2	1	5	51	30.1	168.7
1	2	2	5	8	4.2	4.9
2	2	2	5	7	3.6	4.3
3	2	2	5	8	4.3	4.6
4	2	2	5	8	3.9	4.5
1	2	0	6	89	98.0	1019.6
2	2	0	6	83	64.6	556.9
3	2	0	6	91	96.8	939.4
4	2	0	6	82	81.4	693.5
1	2	1	6	66	49.8	345.9
2	2	1	6	61	55.6	375.5
3	2	1	6	70	54.5	402.2
4	2	1	6	76	72.8	574.8
1	2	2	6	44	25.3	128.0
2	2	2	6	40	21.8	112.6
3	2	2	6	44	26.1	121.1
4	2	2	6	42	23.9	117.9
1	2	0	7	87	92.9	934.2
2	2	0	7	84	66.6	568.9
3	2	0	7	93	107.8	1058.6
4	2	0	7	83	83.9	708.4
1	2	1	7	73	65.4	500.5
2	2	1	7	68	73.3	541.1
3	2	1	7	74	72.1	565.0
4	2	1	7	82	96.2	824.6
1	2	2	7	60	46.3	295.8

Table A.2 Continued.

Block	Year	Burn	Month	Height (cm)	Area (dm^2)	Volume (dm^3)
2	2	2	7	60	40.2	264.8
3	2	2	7	62	42.9	283.5
4	2	2	7	59	40.2	254.6
1	2	0	8	87	100.4	1009.5
2	2	0	8	85	79.0	697.0
3	2	0	8	93	100.0	982.9
4	2	0	8	84	96.9	827.9
1	2	1	8	72	70.5	526.4
2	2	1	8	69	70.4	528.7
3	2	1	8	75	69.3	539.3
4	2	1	8	83	100.0	867.3
1	2	2	8	61	47.6	312.5
2	2	2	8	61	49.0	324.8
3	2	2	8	59	44.0	280.4
4	2	2	8	57	41.2	251.9
1	2	0	9	88	99.0	1041.8
2	2	0	9	85	88.9	788.1
3	2	0	9	92	100.0	971.3
4	2	0	9	84	100.1	866.9
1	2	1	9	71	66.5	499.0
2	2	1	9	70	78.5	606.7
3	2	1	9	74	71.6	552.4
4	2	1	9	80	98.6	812.8
1	2	2	9	60	53.1	343.2
2	2	2	9	60	53.6	349.3
3	2	2	9	60	49.2	315.8
4	2	2	9	59	48.9	304.2

Table A.3. Grass, forb, and total herbage standing crop by year, block, and burn treatment collected in Sep. 2000 and Sep. 2001 on the Hal and Fern Cooper Wildlife Management Area, Fort Supply, OK.

Year	Block	Burn	Distance	Grass	Forb	Total
			-- (m) --	------------	kg ha^{-1}	------------
1	1	1	0	527	3,470	3,997
1	2	1	0	575	1,663	2,238
1	3	1	0	389	1,184	1,573
1	4	1	0	536	1,148	1,684
1	1	2	0	592	418	1,010
1	2	2	0	384	1,257	1,641
1	3	2	0	616	1,233	1,849
1	4	2	0	549	953	1,501
1	1	1	50	2,139	1,257	3,396
1	2	1	50	1,113	923	2,036
1	3	1	50	3,751	471	4,222
1	4	1	50	1,819	766	2,585
1	1	2	50	2,461	471	2,932
1	2	2	50	2,250	1,370	3,619
1	3	2	50	1,756	521	2,277
1	4	2	50	3,136	1,026	4,162
1	1	1	100	4,006	156	4,162
1	2	1	100	2,467	792	3,259
1	3	1	100	4,010	460	4,469
1	4	1	100	2,168	971	3,139
1	1	2	100	3,653	317	3,970
1	2	2	100	2,348	1,172	3,520
1	3	2	100	3,213	206	3,419
1	4	2	100	2,827	1,316	4,143
1	1	1	200	5,348	175	5,522
1	2	1	200	2,586	751	3,337
1	3	1	200	5,149	129	5,278
1	4	1	200	1,813	1,403	3,216
1	1	2	200	4,110	217	4,326
1	2	2	200	3,147	648	3,795
1	3	2	200	4,734	647	5,381
1	4	2	200	1,178	954	2,132
1	1	1	400	4,405	692	5,097
1	2	1	400	3,021	1,054	4,075
1	3	1	400	2,865	687	3,552
1	4	1	400	4,429	871	5,300
1	1	2	400	4,203	146	4,350

Table A.3 Continued.

Year	Block	Burn	Distance	Grass	Forb	Total
			-- (m) --	------------	kg ha^{-1}	------------
1	2	2	400	4,007	913	4,920
1	3	2	400	4,061	862	4,923
1	4	2	400	3,143	644	3,786
1	1	1	800	4,776	475	5,251
1	2	1	800	5,106	658	5,764
1	3	1	800	3,264	262	3,527
1	4	1	800	2,259	909	2,996
1	1	2	800	4,017	289	4,306
1	2	2	800	5,929	586	6,514
1	3	2	800	6,145	1,015	7,159
1	4	2	800	1,131	2,055	3,186
2	1	1	0	247	382	629
2	2	1	0	257	557	814
2	3	1	0	134	1,782	1,916
2	4	1	0	319	511	830
2	1	2	0	147	544	691
2	2	2	0	293	799	1,091
2	3	2	0	128	1,394	1,522
2	4	2	0	141	221	362
2	1	1	50	897	796	1,693
2	2	1	50	1,393	792	2,185
2	3	1	50	736	206	942
2	4	1	50	1,792	217	2,009
2	1	2	50	1,950	928	2,878
2	2	2	50	1,057	1,316	2,373
2	3	2	50	527	533	1,060
2	4	2	50	936	547	1,484
2	1	1	100	502	650	1,152
2	2	1	100	402	932	1,333
2	3	1	100	602	629	1,230
2	4	1	100	1,564	377	1,941
2	1	2	100	1,835	450	2,285
2	2	2	100	1,354	611	1,964
2	3	2	100	999	666	1,665
2	4	2	100	1,145	326	1,520
2	1	1	200	1,786	1,076	2,862
2	2	1	200	2,130	307	2,437
2	3	1	200	1,928	473	2,401
2	4	1	200	3,264	317	3,581

Table A.3 Continued.

Year	Block	Burn	Distance	Grass	Forb	Total
			-- (m) --	------------	kg ha^{-1}	------------
2	1	2	200	1,663	725	2,388
2	2	2	200	1,130	898	2,029
2	3	2	200	1,239	1,191	2,430
2	4	2	200	1,238	658	1,896
2	1	1	400	2,613	496	3,109
2	2	1	400	2,242	908	3,150
2	3	1	400	1,956	343	2,299
2	4	1	400	1,849	190	2,039
2	1	2	400	887	640	1,527
2	2	2	400	4,463	839	5,301
2	3	2	400	2,167	1,054	3,221
2	4	2	400	2,905	165	3,070
2	1	1	800	3,591	216	3,808
2	2	1	800	3,416	440	3,857
2	3	1	800	2,044	1,465	3,509
2	4	1	800	2,624	685	3,308
2	1	2	800	2,358	362	2,720
2	2	2	800	2,900	765	3,664
2	3	2	800	2,888	164	3,052
2	4	2	800	3,123	352	3,475

Table A.4. Grasshopper abundance per 150 sweeps by burn, year, and species in 2000 and 2001 on the Hal and Fern Cooper Wildlife Management Area, Fort Supply, OK.

Burn	Yr	Agde	Arsi	Arxa	Chvi	Hevi	Hioc	Melbil	Melbiv	Mebo	Merbiv	Opob	Psam	Scob	Sper
		(Individuals per 150 sweeps)													
0	1	4	0	0	0	5	0	0	0	19	0	0	0	0	0
0	1	5	0	0	0	1	0	0	0	6	1	0	0	0	0
0	1	2	0	0	0	1	0	0	0	10	1	0	0	0	0
0	1	1	0	0	0	1	1	0	0	0	2	0	0	0	0
0	2	2	0	0	0	1	2	0	0	3	0	0	0	0	0
0	2	1	0	0	0	0	0	1	1	4	0	0	0	0	0
0	2	2	0	1	0	3	0	0	0	4	0	0	0	0	0
0	2	3	0	0	0	1	0	0	1	8	0	0	0	0	0
1	1	3	0	0	0	1	0	0	0	10	1	0	0	0	2
1	1	0	0	0	0	0	0	0	1	7	1	0	0	0	0
1	1	0	0	1	0	0	0	0	0	6	3	0	0	0	1
1	1	1	0	0	0	0	2	0	0	11	1	0	0	0	0
1	2	3	0	0	1	0	0	0	0	6	0	0	0	0	0
1	2	0	0	1	0	0	1	0	0	4	0	0	0	0	0
1	2	0	0	0	0	0	0	0	0	6	0	0	0	0	0
1	2	0	0	0	0	0	0	0	0	3	0	0	0	0	0
2	1	7	0	0	0	0	0	0	0	5	0	0	0	0	1
2	1	1	0	0	0	0	0	0	0	0	1	0	0	1	0
2	1	2	0	0	0	0	0	0	0	10	0	0	0	0	0
2	1	0	0	0	0	2	0	0	0	9	0	0	1	0	1
2	2	1	0	0	0	0	2	0	0	5	1	1	0	0	0
2	2	0	0	1	0	0	1	0	0	7	0	0	0	0	0
2	2	0	1	0	0	0	0	0	0	2	1	0	0	0	0
2	2	1	0	0	0	0	0	0	0	6	1	0	1	0	1

Table A.5 Grasshopper abundance and biomass in July 2000 and August 2001 by block, burn, and year on the Hal and Fern Cooper Wildlife Management Area, Fort Supply, OK.

Block	Burn	Year	Abundance (#/150 sweeps)	Biomass (mg/150 sweeps)
1	0	1	28	10,556
2	0	1	13	3,848
3	0	1	14	4,760
4	0	1	5	1,700
1	0	2	8	4,648
2	0	2	7	3,605
3	0	2	10	4,590
4	0	2	13	4,537
1	1	1	17	6,528
2	1	1	9	4,311
3	1	1	11	4,026
4	1	1	15	7,155
1	1	2	10	3,220
2	1	2	6	2,484
3	1	2	6	2,496
4	1	2	3	1,194
1	2	1	13	3,549
2	2	1	3	1,287
3	2	1	12	4,224
4	2	1	13	5,642
1	2	2	10	4,680
2	2	2	9	4,590
3	2	2	4	1,196
4	2	2	10	3,380